

November 10, 2003

Risk Assessment Technical Background Document for the Dye and Pigment Industry Hazardous Waste Listing Determination

Prepared for

**Office of Solid Waste
U.S. Environmental Protection Agency
1200 Pennsylvania Avenue NW
Washington, DC 20460**

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1.0 Introduction

1.1 Background

The U.S. Environmental Protection Agency (EPA) Office of Solid Waste (OSW), as directed by Congress in the Hazardous and Solid Waste Amendments of 1984 (HSWA) to the Resource Conservation and Recovery Act (RCRA), began an investigation of the wastes generated by the dyes and pigments industries. This investigation also was mandated by a 1991 consent decree resulting from litigation brought against the Agency. As a result of the investigation, EPA proposed two earlier listing determinations with regard to wastes from dye and pigment manufacturing, one in 1994 and another in 1999.

On December 22, 1994 (59 *FR* 66071), EPA proposed listing determinations for five wastes; a no-list determination for six wastes; and deferred action on three wastes. On July 23, 1999 (64 *FR* 40192), the Agency proposed listing determinations for the three deferred wastes from the 1994 proposed rule. Neither of these proposals was finalized, however, and EPA is no longer following the approach proposed in these actions. Instead, the Agency is proposing a new approach for all of the wastes at issue, which uses a “mass-loading” concept. Therefore, the prior proposals are no longer relevant to the proposal supported by this background document.

To develop the supporting data for the 1994 and 1999 listing rulemakings for the dye and pigment industry, a questionnaire was sent out to industry pursuant to RCRA Section 3007 and the agency conducted waste sampling and analysis. Some of the information collected from industry was claimed as confidential. As a result of a consent order and a subsequent preliminary injunction in connection with a case brought by some of the pigment manufacturers to prevent the disclosure of information claimed as confidential business information (CBI), *Magruder et al. v. U.S. EPA*, Civ. No. 94-5768 (D.N.J.), EPA is enjoined from disclosing certain information claimed as confidential.

1.2 Purpose of the Risk Assessment

The purpose of this document is to describe EPA’s development of risk-based mass loading limits for constituents contained in dye and pigment waste streams. These mass loading limits are constituent-specific numerical limits that are protective of human health and the environment. They can be defined as the maximum annual mass loads at which adverse health effects from any single constituent do not exceed a 1×10^{-5} (1 in 100,000) individual lifetime cancer risk or a noncancer hazard quotient (HQ) of 1 at the upper end of the risk distribution for any potential human exposure route.

1.3 Overview of Risk Assessment Methodology

This risk assessment provides estimates of allowable chemical mass that can be disposed of annually in a WMU and remain protective of human health and the environment at specified target risk levels. For human health, one of two target risk levels was used in this analysis:

- An estimate of the excess lifetime cancer risk for individuals exposed to carcinogenic (cancer-causing) contaminants of 1 chance in 100,000 (1E-5)

or

- A ratio of projected intake levels to safe intake levels, called a hazard quotient (HQ), of 1 for constituents that can produce noncancer health effects.

Effects to ecological receptors were assessed using a ratio of predicted surface water concentration to ambient water quality criteria, also known as an HQ. The target for the ecological HQ was also 1.

The risk assessment included six primary tasks:

1. Establish the characteristics of the WMUs where wastes may be managed.
2. Construct the environmental setting where WMUs receiving dye and pigment wastes may be located.
3. Identify scenarios under which contaminants are released from a WMU and transported to human or ecological receptors.
4. Predict the fate and transport of constituents in the environment once they are released from a WMU.
5. Quantify an individual's exposure to the contaminant in the environment.
6. Determine the receptor's predicted risk due to the exposure and use the risk information to determine the mass of each constituent that can be safely managed in each WMU.

This risk assessment was conducted using a probabilistic analysis. A probabilistic risk analysis produces a distribution of risk or hazard for each receptor by allowing some of the parameters in the analysis to have more than one value. A probabilistic analysis is ideal for this risk assessment because dye and pigment wastes could be disposed of in a wide variety of sizes and types of WMUs. The probabilistic analysis not only captures the variability in waste management practices, it also captures the differences in environmental settings (e.g., hydrology, meteorology) where dye and pigment wastes may be managed and in exposure parameters, such as receptor distance, ingestion rates, and inhalation rates.

1.4 Document Organization

This background document is organized into the following sections:

- Section 2, Allowable Mass Loadings for Dye and Pigment Constituents, presents a summary of risk assessment results and the allowable mass loading limits that are protective of human health and the environment.
- Section 3, Risk Assessment Overview, presents the conceptual framework for the risk assessment, including a description of waste streams and waste management practices, fate and transport modeling, exposure assessment, and calculation of protective waste and leachate concentrations.
- Section 4, Source Characterization, presents the methodologies used to characterize the environmental setting, including delineation of the site layout and environmental setting (e.g., meteorology, climate, soils, and aquifers). It also describes the characterization of WMUs, including capacities and surface areas.
- Section 5, Exposure Point Concentrations, describes the models and methods used to estimate the concentration to which receptors may be exposed (i.e., source modeling, air dispersion modeling, groundwater modeling, and shower modeling).
- Section 6, Human Exposure Assessment, presents an overview of the human receptors, selected exposure pathways, and exposure scenarios considered for this assessment. It also presents particular exposure factors (i.e., values needed to calculate human exposure) used in the analysis and presents methods used to estimate dose, including average daily dose (ADD) and lifetime average daily dose (LADD).
- Section 7, Human Health Toxicity Assessment, presents the noncancer and cancer benchmarks used to evaluate human health effects that may result from exposure to constituents modeled for this risk assessment.
- Section 8, Human Health Risk Estimation, presents the methods used to characterize the risk posed to an individual.
- Section 9, Allowable Mass Loading Calculations, describes the calculation methods used to generate the allowable mass loadings.
- Section 10, Variability and Uncertainty, discusses the methods that were used in the dyes and pigments listing risk assessment to account for variability and uncertainty.

The following appendices, A through I, provide supplemental technical information and supporting data:

- Appendix A – Allowable Mass Loadings
- Appendix B – Chemical-Specific Parameters for Source Partitioning and Fate and Transport Models
- Appendix C – Distribution Coefficients
- Appendix D – Source Modeling Parameters
- Appendix E – Groundwater Modeling Parameters
- Appendix F – Unitized Air Concentrations
- Appendix G – Exposure Equations
- Appendix H – Human Exposure Factors
- Appendix I – Model Validation
- Appendix J – Groundwater Time to Impact

2.0 Allowable Mass Loadings for Dye and Pigment Constituents

This section provides a summary of allowable mass loadings calculated for dye and pigment constituents. These protective mass loadings were established to ensure that 90 percent of the time target risk levels of 1 in 100,000 (1×10^{-5}) individual lifetime cancer risk or HQs of 1.0 for noncancer health effects are not exceeded for the waste management scenarios evaluated. Protective mass loadings were also established to ensure that 90 percent of the time, surface water standards would not be exceeded. Surface water standards were evaluated using a screening approach to assess the groundwater-to-surface-water pathway.

An overview of the assessment on which these results are based (e.g., waste management scenarios, analysis framework) is provided in Section 3.0. Other sections of this report provide more details on analysis methodologies, parameter values, assumptions, and uncertainties. Section 2-1 presents results from the human health risk assessment and the ecological screening. Tables summarizing these results are presented at the end of this section. Appendix A provides more detailed results.

2.1 Mass Loadings Based on Human Health

This section presents the mass loadings that have been determined to be protective of human health 90 percent of the time at the target risk levels. Results are presented for three WMUs: landfills, surface impoundments, and tanks.

Probabilistic results are based on a Monte Carlo simulation in which many model input parameter values are varied over 10,000 iterations of the model to yield a statistical distribution of exposures and risks. Results shown in this section are based on the 90th percentile level identified from these distributions. The 90th percentile level means that the mass loading listed in the result tables will be protective of human health at the specified target risk level for 90 percent of the scenarios in the Monte Carlo analysis. Results at other percentile levels are presented in Appendix A.

Several tables are presented at the end of this section for each WMU that provide mass loads for different scenarios and assumptions. These tables are first organized by WMU such that all the results for landfills, surface impoundments, and tanks are group together. The tables are further divided depending on whether or not biodegradation was considered for organic chemicals. For each WMU, the first of tables include all chemicals of concern and assume that all organic chemicals are biologically degraded. The majority of organic chemicals considered had degradation rates available from an agency approved source; however, for chemicals without

degradation rates, a surrogate chemical was selected as the source of degradation rates used to generate these results. (See Appendix B for further details of the surrogate selection process). For comparison purposes, the next set of tables provides mass loads for organic chemicals for which surrogates were selected and using a degradation rate of zero. The final level of grouping in the result tables is based on the source of emissions. For landfills and surface impoundments, mass loads are presented in separated tables which are based on above-ground and groundwater emissions. Since tanks were assumed to have an impervious bottom, only above-ground emissions are estimated for tanks so this last level of differentiating the results is not needed.

Additionally, the tables show the endpoint and cohort that are the basis for the protective mass levels. For example, if the 90th percentile mass load for a given constituent indicates that cancer was the limiting endpoint and the adult was the limiting receptor, this means that cancer risk to an adult receptor had the lowest 90th percentile mass load for the endpoints and receptors evaluated for a given waste management scenario and constituent. The tables also note the driving pathway as being either inhalation, drinking water, or shower. Adult receptors can be exposed to constituents in both drinking water and shower air when constituents have both ingestion and inhalation cancer benchmarks. Mass loadings that are based on more than one pathway are denoted by an asterisk. Results for other receptors and pathways are presented in Appendix A.

2.1.1 Mass Loadings for Landfills

Tables 2-1 and 2-2 present the mass loadings for landfills that, when used in the Monte Carlo analysis, yield a distribution of risk levels where the 90th percentile is equal to the target risk level. Table 2-1 presents results for all chemicals and assuming that all organic chemicals are biodegraded to some degree. Table 2-2 presents results without degradation for the organic chemicals in that had degradation rates based on surrogate chemicals as described above in Section 2.1. Tables 2-1a and 2-2a present the mass loadings due to aboveground emissions. Tables 2-1b and 2-2b present the mass loadings due to groundwater emission.

The analyses conducted for this risk assessment included three scenarios for landfills: no liner (NL), clay liner (CL), and synthetic liner (SL). An additional scenario was also considered for composite liners (GM-GCL). The modeling assumed a lifetime of 30 years and used a distribution of 100 percent municipal solid waste landfills.

2.1.2 Mass Loadings for Surface Impoundments

Tables 2-3 and 2-4 present the mass loadings for surface impoundments that when used in the Monte Carlo analysis, yielded a distribution of risk levels with a target risk at the 90th percentile level. Table 2-3 presents results assuming all organic constituents degrade and Table 2-4 presents results assuming that certain constituents do not degrade. The mass loadings due to aboveground and groundwater emissions are presented separately. The results for surface impoundments are presented for three liner scenarios: no liner (NL), clay liner (CL), and composite liner (COMP).

2.1.3 Mass Loadings for Tanks

Tables 2-5 and 2-6 present the mass loadings for tanks that, when used in the Monte Carlo analysis, yielded a distribution of risk levels with a target risk at the 90th percentile level. Table 2-5 presents results assuming all organic constituents degrade and Table 2-6 presents results assuming that certain constituents do not degrade. Tanks are not considered as a source of groundwater contamination and therefore only have risks due to aboveground emissions.

2.2 Groundwater to Surface Water Screening

Tables 2-7 and 2-8 present the mass loadings that have been determined to be protective of surface water 90 percent of the time. The surface water assessment used a screening level approach to estimate concentrations in surface water based on the groundwater to surface water pathway.

Table 2-1a. Minimum 90th Percentile Allowable Mass Loadings in Landfills Due to Aboveground Emissions and Including Degradation

CAS	Chemical	Mass Loading in kg/yr			
		NL	CL	SL	GM-GCL
62-53-3	Aniline	3.8E+04 (N-C-I)	4.2E+04 (N-C-I)	3.4E+04 (N-C-I)	2.5E+04 (N-C-I)
90-04-0	o-Anisidine	5.9E+04 (C-C-I)	5.6E+04 (C-C-I)	3.3E+04 (C-C-I)	2.4E+04 (C-C-I)
50-00-0	Formaldehyde	6.4E+04 (C-C-I)	5.8E+04 (C-C-I)	2.8E+04 (C-C-I)	2.0E+04 (C-A-I)
95-53-4	o-Toluidine	2.2E+05 (C-C-I)	2.3E+05 (C-C-I)	2.1E+05 (C-A-I)	1.4E+05 (C-C-I)
91-20-3	Naphthalene	2.4E+05 (N-C-I)	2.5E+05 (N-C-I)	2.3E+05 (N-C-I)	1.7E+05 (N-C-I)
95-80-7	Toluene-2,4-diamine	2.6E+05 (C-A-I)	3.4E+05 (C-C-I)	9.2E+04 (C-A-I)	5.0E+04 (C-A-I)
120-71-8	p-Cresidine	4.0E+05 (C-A-I)	3.9E+05 (C-A-I)	2.3E+05 (C-C-I)	1.6E+05 (C-C-I)
103-33-3	Azobenzene	5.1E+05 (C-C-I)	4.6E+05 (C-A-I)	3.0E+05 (C-A-I)	2.3E+05 (C-A-I)
92-87-5	Benzidine	5.3E+05 (C-C-I)	8.3E+05 (C-C-I)	1.6E+05 (C-C-I)	7.7E+04 (C-A-I)
95-50-1	1,2-Dichlorobenzene	1.3E+06 (N-C-I)	1.2E+06 (N-C-I)	8.2E+05 (N-C-I)	6.3E+05 (N-C-I)
91-94-1	3,3-Dichlorobenzidine	8.4E+06 (C-C-I)	1.0E+07 (C-C-I)	2.9E+06 (C-C-I)	1.6E+06 (C-C-I)
99-59-2	5-Nitro-o-anisidine	1.2E+07 (C-C-I)	1.1E+07 (C-C-I)	4.2E+06 (C-C-I)	3.0E+06 (C-A-I)
67-56-1	Methanol	2.0E+07 (N-C-I)	1.8E+07 (N-C-I)	1.1E+07 (N-C-I)	8.2E+06 (N-C-I)
101-77-9	4,4-Methylenedianiline	6.9E+07 (C-C-I)	1.1E+08 (C-C-I)	2.1E+07 (C-C-I)	1.0E+07 (C-A-I)
106-44-5	p-Cresol	1.7E+08 (N-C-I)	1.8E+08 (N-C-I)	1.4E+08 (N-C-I)	1.0E+08 (N-C-I)
117-79-3	2-Aminoanthraquinone	5.3E+08 (C-C-I)	8.0E+08 (C-C-I)	1.7E+08 (C-C-I)	8.3E+07 (C-A-I)
90-94-8	4-4-Bis(dimethylamino) benzophenone	8.1E+09 (C-C-I)	1.2E+10 (C-C-I)	2.4E+09 (C-C-I)	1.1E+09 (C-A-I)

* Result is based on more than one pathway. A description of codes follows Table 2-8.

Table 2-1b. Minimum 90th Percentile Allowable Mass Loadings in Landfills Due to Groundwater Emissions and Including Degradation

CAS	Chemical	Mass Loading in kg/yr			
		NL	CL	SL	GM-GCL
95-80-7	Toluene-2,4-diamine	3.4E-01 (C-A-D)*	9.9E-01 (C-A-D)*	1.4E+02 (C-A-D)*	3.4E+10 (C-A-D)*
95-68-1	2,4-Dimethylaniline (2,4-xylydine)	2.1E+01 (C-A-D)	1.0E+02 (C-A-D)	1.7E+08 (C-A-D)	1.2E+12 (C-A-D)
90-04-0	o-Anisidine	3.0E+01 (C-A-D)*	1.1E+02 (C-A-D)*	2.9E+05 (C-A-D)*	1.4E+12 (C-A-D)*
7632-00-0	Sodium Nitrite	1.0E+02 (N-C-D)	1.5E+02 (N-C-D)	4.0E+03 (N-C-D)	1.9E+06 (N-C-D)
92-87-5	Benzidine	1.2E+02 (C-A-D)*	4.3E+06 (C-A-D)*	3.3E+09 (C-A-D)*	1.4E+11 (C-A-D)*
120-71-8	p-Cresidine	1.2E+02 (C-A-D)*	6.6E+02 (C-A-D)*	3.4E+09 (C-C-D)	5.0E+12 (C-A-D)*
95-54-5	1,2-Phenylenediamine	1.6E+02 (C-A-D)	7.1E+02 (C-A-D)	1.0E+07 (C-A-D)	4.9E+13 (C-A-D)
108-45-2	1,3-Phenylenediamine	3.0E+02 (N-C-D)	1.2E+03 (N-C-D)	6.1E+06 (N-C-D)	9.3E+14 (N-C-D)
106-47-8	4-Chloroaniline	6.3E+02 (C-A-D)	4.8E+03 (C-A-D)	6.8E+10 (C-A-D)	2.6E+14 (C-A-D)
7439-92-1	Lead	1.3E+03 (M-M-D)	4.9E+03 (M-M-D)	3.5E+05 (M-M-D)	3.0E+09 (M-M-D)
62-53-3	Aniline	1.9E+03 (C-A-D)*	9.3E+03 (C-A-D)*	1.4E+09 (C-A-D)*	2.5E+14 (C-A-D)*
121-69-7	N,N-Dimethylaniline	2.5E+03 (N-C-D)	6.5E+04 (N-C-D)	5.0E+11 (N-C-D)	2.4E+13 (N-C-D)
106-50-3	1,4-Phenylenediamine	6.5E+03 (N-C-D)	2.4E+04 (N-C-D)	4.7E+07 (N-C-D)	2.5E+15 (N-C-D)
7440-50-8	Copper	1.5E+04 (M-M-D)	5.2E+04 (M-M-D)	2.0E+06 (M-M-D)	9.8E+09 (M-M-D)
50-00-0	Formaldehyde	3.6E+04 (N-C-D)	2.3E+05 (N-C-D)	8.0E+11 (N-C-D)	2.4E+15 (N-C-D)
101-77-9	4,4-Methylenedianiline	3.6E+04 (C-C-D)	3.6E+09 (C-A-D)*	5.7E+11 (C-A-D)*	2.4E+13 (C-A-D)*
100-52-7	Benzaldehyde	4.5E+04 (N-C-D)	5.2E+05 (N-C-D)	1.4E+13 (N-C-D)	1.4E+15 (N-C-D)
119-90-4	3,3-Dimethoxybenzidine	9.1E+04 (C-A-D)	2.3E+07 (C-A-D)	2.4E+13 (C-A-D)	2.0E+16 (C-C-D)
7440-66-6	Zinc	9.3E+04 (N-C-D)	3.4E+05 (N-C-D)	1.7E+07 (N-C-D)	1.0E+11 (N-C-D)
7440-39-3	Barium	9.4E+04 (N-C-D)	3.0E+05 (N-C-D)	1.4E+07 (N-C-D)	9.4E+10 (N-C-D)
95-53-4	o-Toluidine	1.8E+05 (C-A-D)*	1.2E+10 (C-A-D)*	2.6E+12 (C-A-D)*	9.6E+13 (C-A-D)*
106-49-0	p-Toluidine	2.3E+05 (C-A-D)	1.5E+10 (C-A-D)	3.0E+12 (C-A-D)	1.7E+15 (C-A-D)
67-56-1	Methanol	2.9E+06 (N-C-D)	3.5E+08 (N-C-D)	2.4E+14 (N-C-D)	4.7E+15 (N-C-D)
108-95-2	Phenol	8.4E+07 (N-C-D)	5.9E+12 (N-C-D)	1.5E+15 (N-C-D)	5.2E+16 (N-C-D)
99-59-2	5-Nitro-o-anisidine	7.3E+08 (C-C-D)	6.4E+09 (C-A-D)*	1.5E+13 (C-A-D)*	8.7E+14 (C-A-D)*
95-50-1	1,2-Dichlorobenzene	1.0E+10 (N-A-S)	4.6E+11 (N-A-S)	4.3E+12 (N-A-S)	1.2E+14 (N-A-S)
91-94-1	3,3-Dichlorobenzidine	2.2E+10 (C-A-D)*	2.4E+11 (C-A-D)*	6.1E+12 (C-A-D)*	8.9E+14 (C-A-D)*
106-44-5	p-Cresol	4.1E+10 (N-C-D)	4.2E+12 (N-C-D)	5.5E+13 (N-C-D)	2.1E+15 (N-C-D)
119-93-7	3,3-Dimethylbenzidine	8.7E+10 (C-A-D)	2.5E+11 (C-A-D)	3.7E+12 (C-A-D)	2.6E+17 (C-A-D)
91-20-3	Naphthalene	2.3E+11 (N-A-S)	8.9E+11 (N-A-S)	1.2E+13 (N-A-S)	7.1E+14 (N-A-S)
117-79-3	2-Aminoanthraquinone	4.2E+11 (C-A-D)*	1.5E+12 (C-A-D)*	2.8E+13 (C-A-D)*	2.1E+15 (C-A-D)*
99-55-8	5-Nitro-o-toluidine	1.4E+12 (C-A-D)	6.1E+12 (C-A-D)	5.6E+13 (C-A-D)	2.4E+15 (C-A-D)
90-94-8	4-4-Bis(dimethylamino) benzophenone	2.1E+12 (C-A-D)*	5.6E+12 (C-A-D)*	3.3E+14 (C-A-D)*	2.4E+17 (C-A-D)*
103-33-3	Azobenzene	3.5E+12 (C-A-D)*	8.0E+12 (C-A-D)*	1.4E+14 (C-A-D)*	1.4E+16 (C-A-D)*
122-39-4	Diphenylamine	7.1E+13 (N-C-D)	2.0E+14 (N-C-D)	5.2E+15 (N-C-D)	4.6E+26 (N-C-D)

* Result is based on more than one pathway. A description of codes follows Table 2-8.

Table 2-2a. Minimum 90th Percentile Allowable Mass Loadings in Landfills Due to Aboveground Emissions with No Degradation

CAS	Chemical	Mass Loading in kg/yr			
		NL	CL	SL	GM-GCL
62-53-3	Aniline	7.3E+03 (N-C-I)	7.6E+03 (N-C-I)	6.3E+03 (N-C-I)	4.9E+03 (N-C-I)
103-33-3	Azobenzene	9.7E+04 (C-A-I)	8.2E+04 (C-A-I)	5.2E+04 (C-A-I)	4.0E+04 (C-A-I)

* Result is based on more than one pathway. A description of codes follows Table 2-8.

Table 2-2b. Minimum 90th Percentile Allowable Mass Loadings in Landfills Due to Groundwater Emissions with No Degradation

CAS	Chemical	Mass Loading in kg/yr			
		NL	CL	SL	GM-GCL
95-68-1	2,4-Dimethylaniline (2,4-xylydine)	1.2E+00 (C-A-D)	3.7E+00 (C-A-D)	1.6E+02 (C-A-D)	1.2E+05 (C-A-D)
95-54-5	1,2-Phenylenediamine	3.0E+00 (C-A-D)	5.7E+00 (C-A-D)	1.8E+02 (C-A-D)	1.1E+05 (C-A-D)
106-49-0	p-Toluidine	3.8E+00 (C-A-D)	1.1E+01 (C-A-D)	4.0E+02 (C-A-D)	3.3E+06 (C-A-D)
106-47-8	4-Chloroaniline	2.9E+01 (C-A-D)	8.9E+01 (C-A-D)	3.4E+03 (C-A-D)	6.3E+07 (C-A-D)
62-53-3	Aniline	4.4E+01 (C-A-D)*	1.1E+02 (C-A-D)*	4.3E+03 (C-A-D)*	3.2E+06 (C-A-D)*
100-52-7	Benzaldehyde	5.5E+02 (N-C-D)	1.5E+03 (N-C-D)	7.1E+04 (N-C-D)	2.2E+08 (N-C-D)
103-33-3	Azobenzene	1.5E+03 (C-A-D)*	6.8E+03 (C-A-D)*	2.4E+06 (C-A-D)*	3.6E+10 (C-A-D)*

* Result is based on more than one pathway. A description of codes follows Table 2-8.

**Table 2-3a. Minimum 90th Percentile Allowable Mass Loadings in
Surface Impoundments Due to Aboveground Emissions and Including Degradation**

CAS	Chemical	Mass Loading in kg/yr		
		NL	CL	COMP
62-53-3	Aniline	1.5E+03 (N-C-I)	1.5E+03 (N-C-I)	1.5E+03 (N-C-I)
91-20-3	Naphthalene	2.2E+03 (N-C-I)	2.2E+03 (N-C-I)	2.2E+03 (N-C-I)
95-53-4	o-Toluidine	2.4E+03 (C-C-I)	2.3E+03 (C-C-I)	2.4E+03 (C-C-I)
103-33-3	Azobenzene	2.4E+03 (C-C-I)	2.4E+03 (C-C-I)	2.4E+03 (C-C-I)
90-04-0	o-Anisidine	3.0E+03 (C-A-I)	2.9E+03 (C-A-I)	2.9E+03 (C-A-I)
120-71-8	p-Cresidine	1.3E+04 (C-A-I)	1.3E+04 (C-A-I)	1.3E+04 (C-A-I)
50-00-0	Formaldehyde	1.4E+04 (C-C-I)	1.4E+04 (C-C-I)	1.4E+04 (C-C-I)
95-80-7	Toluene-2,4-diamine	5.5E+04 (C-A-I)	5.1E+04 (C-A-I)	5.1E+04 (C-A-I)
95-50-1	1,2-Dichlorobenzene	6.3E+04 (N-C-I)	6.3E+04 (N-C-I)	6.3E+04 (N-C-I)
92-87-5	Benzidine	9.0E+04 (C-A-I)	8.9E+04 (C-A-I)	8.9E+04 (C-A-I)
91-94-1	3,3-Dichlorobenzidine	1.2E+05 (C-A-I)	1.2E+05 (C-A-I)	1.2E+05 (C-A-I)
99-59-2	5-Nitro-o-anisidine	1.1E+06 (C-A-I)	1.1E+06 (C-A-I)	1.1E+06 (C-A-I)
67-56-1	Methanol	3.4E+06 (N-C-I)	3.4E+06 (N-C-I)	3.4E+06 (N-C-I)
106-44-5	p-Cresol	3.8E+06 (N-C-I)	3.8E+06 (N-C-I)	3.8E+06 (N-C-I)
101-77-9	4,4-Methylenedianiline	1.1E+07 (C-A-I)	1.1E+07 (C-A-I)	1.1E+07 (C-A-I)
90-94-8	4,4-Bis(dimethylamino) benzophenone	2.5E+08 (C-A-I)	2.4E+08 (C-A-I)	2.4E+08 (C-A-I)
117-79-3	2-Aminoanthraquinone	2.5E+08 (C-A-I)	2.4E+08 (C-A-I)	2.5E+08 (C-A-I)

* Result is based on more than one pathway. A description of codes follows Table 2-8.

**Table 2-3b. Minimum 90th Percentile Allowable Mass Loadings in
Surface Impoundments Due to Groundwater Emissions and Including Degradation**

CAS	Chemical	Mass Loading in kg/yr		
		NL	CL	COMP
92-87-5	Benzidine	7.1E-02 (C-A-D)*	1.8E-01 (C-A-D)*	1.8E+05 (C-A-D)*
95-80-7	Toluene-2,4-diamine	2.1E+00 (C-A-D)*	5.2E+00 (C-A-D)*	3.7E+06 (C-A-D)*
101-77-9	4,4-Methylenedianiline	1.1E+01 (C-A-D)*	2.8E+01 (C-A-D)*	2.8E+07 (C-A-D)*
95-68-1	2,4-Dimethylaniline (2,4-xylydine)	1.8E+01 (C-A-D)	4.4E+01 (C-A-D)	4.1E+07 (C-A-D)
119-93-7	3,3-Dimethylbenzidine	4.1E+01 (C-A-D)	1.2E+02 (C-A-D)	1.2E+10 (C-A-D)
120-71-8	p-Cresidine	5.9E+01 (C-A-D)*	1.5E+02 (C-A-D)*	1.2E+08 (C-A-D)*
95-53-4	o-Toluidine	7.5E+01 (C-A-D)*	1.9E+02 (C-A-D)*	1.9E+08 (C-A-D)*
90-04-0	o-Anisidine	7.6E+01 (C-A-D)*	1.9E+02 (C-A-D)*	1.1E+08 (C-A-D)*
106-49-0	p-Toluidine	9.5E+01 (C-A-D)	2.4E+02 (C-A-D)	5.8E+08 (C-A-D)
95-54-5	1,2-Phenylenediamine	1.8E+02 (C-A-D)	4.5E+02 (C-A-D)	3.2E+08 (C-A-D)
106-47-8	4-Chloroaniline	1.9E+02 (C-A-D)	5.0E+02 (C-A-D)	1.0E+09 (C-A-D)
121-69-7	N,N-Dimethylaniline	2.2E+02 (N-C-D)	5.8E+02 (N-C-D)	6.2E+08 (N-C-D)
91-94-1	3,3-Dichlorobenzidine	2.6E+02 (C-A-D)*	8.3E+02 (C-A-D)*	5.3E+10 (C-A-D)*
108-45-2	1,3-Phenylenediamine	3.9E+02 (N-C-D)	9.7E+02 (N-C-D)	1.3E+09 (N-C-D)
99-59-2	5-Nitro-o-anisidine	4.3E+02 (C-A-D)*	1.1E+03 (C-A-D)*	8.3E+08 (C-A-D)*
7439-92-1	Lead	6.2E+02 (M-M-D)	1.6E+03 (M-M-D)	1.4E+10 (M-M-D)
91-20-3	Naphthalene	6.6E+02 (N-A-S)	2.2E+03 (N-A-S)	3.2E+10 (N-A-S)
119-90-4	3,3-Dimethoxybenzidine	7.6E+02 (C-A-D)	2.0E+03 (C-A-D)	4.2E+09 (C-A-D)
99-55-8	5-Nitro-o-toluidine	1.0E+03 (C-A-D)	2.6E+03 (C-A-D)	3.3E+09 (C-A-D)
106-44-5	p-Cresol	1.3E+03 (N-C-D)	3.3E+03 (N-C-D)	5.5E+09 (N-C-D)
62-53-3	Aniline	1.7E+03 (C-A-D)*	4.4E+03 (C-A-D)*	3.1E+09 (C-A-D)*
90-94-8	4-4-Bis(dimethylamino) benzophenone	1.8E+03 (C-A-D)*	7.6E+03 (C-A-D)*	2.3E+13 (C-C-D)
117-79-3	2-Aminoanthraquinone	3.7E+03 (C-A-D)*	1.2E+04 (C-A-D)*	2.0E+11 (C-A-D)*
7632-00-0	Sodium Nitrite	3.9E+03 (N-C-D)	9.9E+03 (N-C-D)	5.7E+09 (N-C-D)
7440-50-8	Copper	4.1E+03 (M-M-D)	1.1E+04 (M-M-D)	7.2E+10 (M-M-D)
103-33-3	Azobenzene	9.2E+03 (C-A-D)*	3.6E+04 (C-A-D)*	3.9E+12 (C-A-D)*
7440-39-3	Barium	9.3E+03 (N-C-D)	2.6E+04 (N-C-D)	5.7E+11 (N-C-D)
100-52-7	Benzaldehyde	1.1E+04 (N-C-D)	2.6E+04 (N-C-D)	1.8E+10 (N-C-D)
106-50-3	1,4-Phenylenediamine	1.2E+04 (N-C-D)	3.1E+04 (N-C-D)	2.2E+10 (N-C-D)
7440-66-6	Zinc	1.9E+04 (N-C-D)	5.3E+04 (N-C-D)	7.5E+11 (N-C-D)
50-00-0	Formaldehyde	2.7E+04 (N-C-D)	6.7E+04 (N-C-D)	4.2E+10 (N-C-D)
108-95-2	Phenol	3.9E+04 (N-C-D)	9.6E+04 (N-C-D)	8.6E+10 (N-C-D)
122-39-4	Diphenylamine	6.8E+04 (N-C-D)	2.3E+05 (N-C-D)	3.6E+17 (N-C-D)
95-50-1	1,2-Dichlorobenzene	9.7E+04 (N-A-S)	3.1E+05 (N-A-S)	2.5E+12 (N-A-S)
67-56-1	Methanol	9.7E+04 (N-C-D)	2.5E+05 (N-C-D)	1.5E+11 (N-C-D)

* Result is based on more than one pathway. A description of codes follows Table 2-8.

Table 2-4a. Minimum 90th Percentile Allowable Mass Loadings in Surface Impoundments Due to Aboveground Emissions with No Degradation

CAS	Chemical	Mass Loading in kg/yr		
		NL	CL	COMP
62-53-3	Aniline	1.0E+03 (N-C-I)	9.8E+02 (N-C-I)	9.8E+02 (N-C-I)
103-33-3	Azobenzene	1.7E+03 (C-C-I)	1.7E+03 (C-C-I)	1.7E+03 (C-C-I)

* Result is based on more than one pathway. A description of codes follows Table 2-8.

Table 2-4b. Minimum 90th Percentile Allowable Mass Loadings in Surface Impoundments Due to Groundwater Emissions with No Degradation

CAS	Chemical	Mass Loading in kg/yr		
		NL	CL	COMP
95-68-1	2,4-Dimethylaniline (2,4-xylydine)	1.7E+01 (C-A-D)	4.2E+01 (C-A-D)	1.6E+07 (C-A-D)
106-49-0	p-Toluidine	3.5E+01 (C-A-D)	9.1E+01 (C-A-D)	1.4E+08 (C-A-D)
95-54-5	1,2-Phenylenediamine	1.1E+02 (C-A-D)	2.9E+02 (C-A-D)	1.7E+08 (C-A-D)
106-47-8	4-Chloroaniline	1.5E+02 (C-A-D)	3.8E+02 (C-A-D)	7.8E+08 (C-A-D)
62-53-3	Aniline	1.1E+03 (C-A-D)*	3.0E+03 (C-A-D)*	1.7E+09 (C-A-D)*
103-33-3	Azobenzene	1.2E+03 (C-A-D)*	3.9E+03 (C-A-D)*	3.9E+11 (C-A-D)*
100-52-7	Benzaldehyde	7.3E+03 (N-C-D)	1.8E+04 (N-C-D)	1.2E+10 (N-C-D)

* Result is based on more than one pathway. A description of codes follows Table 2-8.

Table 2-5. Minimum 90th Percentile Allowable Mass Loadings in Tanks Due to Aboveground Emissions and Including Degradation

CAS	Chemical	Mass Loading in kg/yr
91-20-3	Naphthalene	2.2E+03 (N-C-I)
95-53-4	o-Toluidine	2.6E+03 (C-A-I)
62-53-3	Aniline	2.7E+03 (N-C-I)
103-33-3	Azobenzene	3.7E+03 (C-C-I)
90-04-0	o-Anisidine	9.5E+03 (C-C-I)
120-71-8	p-Cresidine	5.0E+04 (C-A-I)
95-50-1	1,2-Dichlorobenzene	7.1E+04 (N-C-I)
95-80-7	Toluene-2,4-diamine	2.6E+05 (C-A-I)
92-87-5	Benzidine	2.9E+05 (C-A-I)
91-94-1	3,3-Dichlorobenzidine	2.9E+05 (C-A-I)
50-00-0	Formaldehyde	2.3E+06 (C-A-I)
99-59-2	5-Nitro-o-anisidine	2.9E+06 (C-A-I)
106-44-5	p-Cresol	5.6E+06 (N-C-I)
67-56-1	Methanol	7.0E+06 (N-C-I)
101-77-9	4,4-Methylenedianiline	3.5E+07 (C-C-I)
90-94-8	4-4-Bis(dimethylamino) benzophenone	4.5E+08 (C-C-I)
117-79-3	2-Aminoanthraquinone	5.7E+08 (C-C-I)

* Result is based on more than one pathway. A description of codes follows Table 2-8.

Table 2-6. Minimum 90th Percentile Allowable Mass Loadings in Tanks Due to Aboveground Emissions with No Degradation

CAS	Chemical	Mass Loading in kg/yr
62-53-3	Aniline	2.0E+03 (N-C-I)
103-33-3	Azobenzene	3.2E+03 (C-C-I)

* Result is based on more than one pathway. A description of codes follows Table 2-8.

Table 2-7. 90th Percentile Allowable Mass Loadings in Landfills Due to the Groundwater-to-Surface-Water Pathway

CAS	Chemical	Mass Loading in kg/yr	
		NL	CL
7440-50-8	Copper	7.0E+04	2.4E+05
7440-66-6	Zinc	1.0E+06	3.4E+06

Table 2-8. 90th Percentile Allowable Mass Loadings in Surface Impoundments Due to the Groundwater-to-Surface-Water Pathway

CAS	Chemical	Mass Loading in kg/yr	
		NL	CL
7440-50-8	Copper	1.9E+04	5.4E+04
7440-66-6	Zinc	2.4E+05	6.7E+05

Legend for all section 2 tables

Codes in parentheses indicate endpoint (C = cancer, N = noncancer, M = maximum contaminant level) - cohort (A = adult, C = child, M = Maximum Contaminant Level) - driving pathway (I = inhalation, D = drinking water, S = shower).

3.0 Risk Assessment Overview

This section describes the conceptual framework for the dyes and pigments listing risk assessment. The objective of this risk assessment approach was to support the calculation of a mass loading limit¹ (in kg/yr) for each WMU and chemical constituent across three different liner types: no liner, clay liner, and composite liner (for which two scenarios were evaluated). Because one of the most important goals for this analysis was to determine the relationship between liner type and mass loading limits, the conceptual framework was designed as a comparative risk analysis. That is, the underlying assumptions and data distributions were consistent within a given WMU simulation with the exception of the liner type. This framework supported mass loading calculations that could be used to quantify the impact of various liner types on risk or hazard.

The remainder of this section provides a brief overview of the dyes and pigments risk assessment framework, describing the waste streams and waste management practices, fate and transport modeling, exposure assessment, the structure of the Monte Carlo analysis, and the calculation of protective allowable mass loadings.

3.1 Source Characterization

3.1.1 Waste Management Scenario

For this assessment, EPA classified dye and pigment manufacturing wastes as either wastewater or nonwastewater. As shown in Table 3-1, wastewaters are assumed to be managed in onsite surface impoundments or tanks. Nonwastewaters are assumed to be managed in offsite municipal landfills. The effects of various liner types on landfills and surface impoundments were considered. Specifically, EPA considered WMUs having no liner (NL), clay liners (CL), and synthetic liners (SL) for landfills and surface impoundments. Additionally, for landfills only, an additional composite liner scenario was evaluated (GM-GCL). Tanks have an impervious bottom; therefore, no liner was considered.

¹The mass loading limit is defined as the annual mass load of a constituent to a WMU above which the human health risk may be unacceptable (i.e., cancer risk greater than 1×10^{-5} or HQ greater than 1).

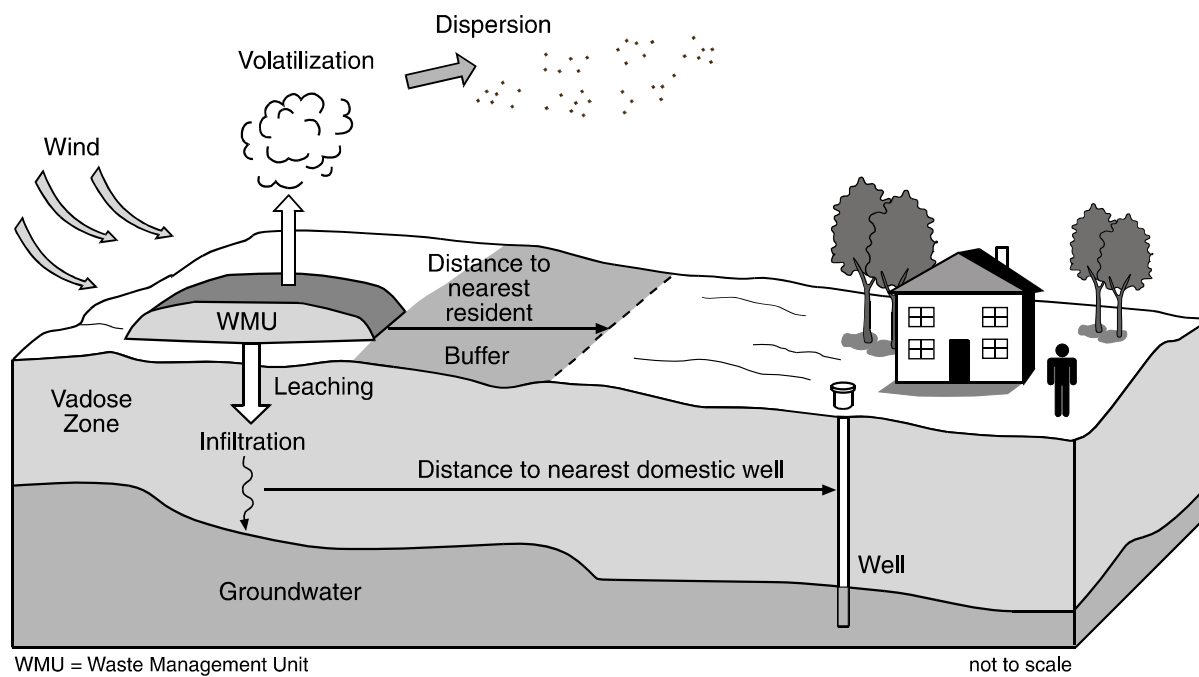
Table 3-1. Waste Management Scenarios Modeled

WMU	Location	Waste Stream
Landfill	Offsite municipal	Nonwastewater
Surface impoundment	Onsite industrial	Wastewater
Tank	Onsite industrial	Wastewater

3.1.2 Site Characterization

The waste stream/WMU combinations discussed above provide the waste management scenarios to be evaluated in the risk assessment. Specific dye and pigment disposal sites were not considered. Rather, WMUs were assumed to be located throughout the contiguous 48 states to reflect nationwide variability in site parameters.

Figure 3-1 depicts the conceptual layout of the sites that were modeled. All receptors are located offsite near the WMU but beyond an intervening area called a buffer area. Beyond the buffer area is a residence. Depending on the release mechanisms for a specific WMU, offsite receptors can come into contact with constituents of concern (COCs) via the ambient air or contaminated groundwater. A screening analysis was also conducted to consider the potential contamination of surface water due to infiltration of groundwater.

**Figure 3-1. Conceptual layout.**

3.1.3 Constituents of Concern

EPA identified COCs associated with dye and pigment waste streams. Table 3-2 below lists the 35 COCs selected; these include four metal constituents and 30 organic constituents. In addition, a screening analysis was conducted for the leaching of sodium nitrite because sorption data were lacking for this chemical (e.g., no K_d data available).

Table 3-2. Dye and Pigment Constituents

CAS	Chemical	Chemical Type
117-79-3	Aminoanthraquinone, 2-	O
62-53-3	Aniline	O
103-33-3	Azobenzene	O
7440-39-3	Barium	M
100-52-7	Benzaldehyde	O
92-87-5	Benzidine	O
90-94-8	Benzophenone, 4-4-bis(dimethylamino)	O
106-47-8	Chloroaniline, 4-	O
7440-50-8	Copper	M
95-50-1	Dichlorobenzene, 1,2-	O
91-94-1	Dichlorobenzidine, 3,3-	O
119-90-4	Dimethoxybenzidine, 3,3-	O
95-68-1	Dimethylaniline, 2,4- (2,4-xylidine)	O
119-93-7	Dimethylbenzidine, 3,3-	O
122-39-4	Diphenylamine	O
50-00-0	Formaldehyde	O
7439-92-1	Lead	M
67-56-1	Methanol	O
101-77-9	Methylenedianiline, 4,4-	O
121-69-7	N,N-Dimethylaniline	O
91-20-3	Naphthalene	O
90-04-0	o-Anisidine	O
99-59-2	o-Anisidine, 5-nitro-	O
95-53-4	o-Toluidine	O
99-55-8	o-Toluidine, 5-nitro-	O
120-71-8	p-Cresidine	O
106-44-5	p-Cresol	O

(continued)

Table 3-2. (continued)

CAS	Chemical	Chemical Type
108-95-2	Phenol	O
95-54-5	Phenylenediamine, 1,2-	O
108-45-2	Phenylenediamine, 1,3-	O
106-50-3	Phenylenediamine, 1,4-	O
106-49-0	p-Toluidine	O
7632-00-0	Sodium nitrite	I
95-80-7	Toluene-2,4-diamine	O
7440-66-6	Zinc	M

Note: O = organic, M = metal, and I = inorganic.

3.2 Exposure Point Concentrations

A series of models was used to estimate concentrations of COCs in the environment with which receptors may come into contact (also termed “exposure point concentrations”). A source model was used to estimate environmental releases of each COC from a WMU for each waste stream. These estimated environmental releases provided input to the fate and transport models to estimate media concentrations in air and groundwater.

3.2.1 Source Modeling

For each WMU type, the release mechanisms to environmental media were evaluated. Table 3-3 lists the primary release mechanisms that apply to each WMU. Volatilization from all three WMUs was considered; however, leaching was considered only from landfills and surface impoundments. Tanks were assumed to have an impervious bottom and, therefore, were not associated with direct releases to the groundwater. Because surface impoundments and tanks contain liquid waste, no particulate emissions are presumed to occur from either of these WMUs. For landfills, it was assumed that erosion and runoff from an operating landfill are controlled, and overland transport of COCs was not included in the source modeling. In addition, the use of a daily cover was assumed to reduce the particulate emissions from landfills to negligible levels; therefore, particulate emissions from landfills were also excluded from this assessment.

Table 3-3. WMU and Primary Release Mechanisms

WMU	Volatilization	Leaching
Landfill	✓	✓
Surface impoundments	✓	✓
Tanks	✓	

3.2.2 Fate and Transport Modeling

As described above, source models were used to determine the amount and nature of constituent released into the environment. Once in the environment, the COCs can move through various environmental media and eventually come into contact with receptors. Table 3-4 lists the environmental media considered for each WMU. Concentrations in ambient air were estimated for organic COCs from all three WMUs, and concentrations in groundwater, for landfills and surface impoundments. For volatile groundwater COCs, the analysis assessed contamination of indoor air due to showering. An additional screening analysis was also conducted to determine if ecological receptors could be affected via the groundwater-to-surface-water pathway.

Table 3-4. WMUs and Affected Environmental Media

WMU	Ambient Air	Groundwater	Indoor Air ¹
Landfill	✓	✓	✓
Surface impoundments	✓	✓	✓
Tanks	✓		

¹ In this analysis, the only source of contamination for indoor air was showering.

Although both aboveground and groundwater pathways were evaluated, they were treated separately in this analysis. This decision was based on differences in the exposure timeframe and receptor location. For most contaminants, a contaminated groundwater plume may not affect a groundwater well for hundreds of years, whereas aboveground exposures to contaminated air may occur simultaneously with the release. In addition, the aboveground receptor locations may not necessarily overlap (i.e., the aboveground receptors are located around the WMU in the predominant wind direction, which may not coincide with the location of the groundwater plume).

3.3 Human Exposure Assessment

The purpose of exposure assessment is to estimate the dose to each receptor type by combining modeled COC concentrations for media with intake rates for those receptors. Table 3-5 lists each receptor, along with the specific exposure pathways that apply to that receptor. Exposure due to inhalation of volatile emissions released above ground and exposure due to chemicals leaching from WMUs and contaminating groundwater were also assessed. The groundwater exposures include tapwater ingestion and inhalation exposures due to showering with tapwater. As noted above, the groundwater pathways were considered separately from the aboveground pathways for the adult resident and the child resident because the timeframe for groundwater exposure may not coincide with the timeframe for ambient inhalation exposure. For the groundwater pathways, it was assumed that the only source of tapwater was well water. In addition, all of the receptors that were considered were assumed to reside offsite at a location near a WMU.

Table 3-5. Receptors and Exposure Pathways

Receptor	Inhalation of Ambient Air	Ingestion of Drinking Water	Inhalation of Indoor Air (Shower) ^b
Adult resident	✓		
Child resident	✓		
Adult resident ^a		✓	✓
Child resident ^a		✓	

^a Groundwater pathways were considered separately from aboveground pathways for the adult resident and the child resident because the timeframe for groundwater exposure is often not consistent with that of other exposure pathways.

^b Showering is explicitly considered for adults and not for children; however, the results can be considered to bracket the risk for this pathway. In terms of noncancer, the hazard quotient is calculated based on the air concentrations, which are the same regardless of age, and the time spent in the shower, which is based on data for all ages. For cancer, adults are typically the most sensitive receptor due to the longer-term exposure.

3.4 Human Health Toxicity Assessment

To characterize chronic risk from human exposures to a COC, toxicity information was identified for use with the exposure assessment results. For this risk assessment, the toxicity of a constituent was defined by cancer and noncancer human health benchmarks for each route of exposure (e.g., inhalation and ingestion). Essentially, a benchmark is a quantitative value used to predict a chemical's possible toxicity and ability to induce a health effect at certain levels of exposure. These health benchmarks are derived from toxicity data based on animal studies or human epidemiological studies. Each benchmark represents a dose-response estimate that relates the likelihood and severity of adverse health effects to exposure and dose. Because individual chemicals cause different health effects at different doses, benchmarks are chemical-specific. Table 3-6 summarizes the types of human health benchmarks used in this risk assessment.

Table 3-6. Chronic Health Benchmarks Used in the Dyes and Pigments Risk Assessment

Constituent	CASRN	RfD	RfC	CSFo	CSFi	MCL
Aminoanthraquinone, 2-	117-79-3			✓	✓	
Aniline	62-53-3		✓	✓	✓	
Anisidine, o-	90-04-0			✓	✓	
Azobenzene	103-33-3			✓	✓	
Barium	7440-39-3	✓	✓			
Benzaldehyde	100-52-7	✓				
Benzidine	92-87-5	✓		✓	✓	

(continued)

Table 3-6. (continued)

Constituent	CASRN	RfD	RfC	CSFo	CSFi	MCL
Bis(dimethylamino) benzophenone, 4-4	90-94-8			✓	✓	
Chloroaniline, 4- (p-)	106-47-8	✓		✓		
Copper	7440-50-8					✓
Cresidine, p-	120-71-8			✓	✓	
Cresol, p- (4-methylphenol)	106-44-5	✓	✓			
Dichlorobenzene, 1,2- (o-)	95-50-1	✓	✓			
Dichlorobenzidine, 3,3'-	91-94-1			✓	✓	
Dimethoxybenzidine, 3,3'-	119-90-4			✓		
Dimethylaniline, 2,4-	95-68-1			✓		
Dimethylbenzidine, 3,3'-	119-93-7			✓		
Diphenylamine, N,N-	122-39-4	✓				
Formaldehyde	50-00-0	✓			✓	
Lead	7439-92-1					✓
Methanol	67-56-1	✓	✓			
Methyl-5-nitroaniline, 2-	99-55-8			✓		
Methylaniline, 2- (o-toluidine)	95-53-4			✓	✓	
Methylene-bisbenzeneamine, 4,4'- (4,4'-methylenedianiline)	101-77-9	✓	✓	✓	✓	
Naphthalene	91-20-3	✓	✓			
Nitro-o-anisidine, 5-	99-59-2			✓	✓	
N-N-Dimethylaniline	121-69-7	✓				
Phenol	108-95-2	✓				
Phenylenediamine, m-	108-45-2	✓				
Phenylenediamine, o-	95-54-5			✓		
Phenylenediamine, p-	106-50-3	✓				
Sodium nitrite	7632-00-0	✓				
Toluene-2,4-diamine (2,4-diaminotoluene)	95-80-7			✓	✓	
Toluidine, p-	106-49-0			✓		
Zinc	7440-66-6	✓				

Key:

- CASRN = Chemical Abstract Service Registry Number
 CSFo = oral cancer slope factor
 CSFi = inhalation cancer slope factor
 RfD = reference dose
 RfC = reference concentration
 MCL = maximum contaminant level
 URF = inhalation unit risk factor.

3.5 Human Health Risk Estimation

Several risk endpoints were used to characterize risk for the human receptors evaluated in this assessment. The term risk endpoint refers to the particular measure of human health hazard or risk (e.g., lifetime excess cancer risk). Table 3-7 lists the risk endpoints used in this risk assessment.

A risk endpoint is a specific type of risk estimate (e.g., individual cancer risk estimate) that is used as the metric for a given risk category. The dyes and pigments listing risk assessment evaluated two types of health effects: cancer effects and noncancer effects. Each of the COCs was evaluated for one or both of these health effects depending on the available toxicological benchmarks.

Table 3-7. Risk Endpoints for Cancer and Noncancer Effects

Risk Category	Risk Endpoint	Definition
Cancer effects	Lifetime excess cancer risk - inhalation	Lifetime excess cancer risk resulting from inhalation exposure to a single chemical
	Lifetime excess cancer risk - ingestion	Lifetime excess cancer risk resulting from ingestion exposure to a single chemical
	Total lifetime excess cancer risk	Lifetime excess cancer risk resulting from multiple pathway exposures to a single chemical (inhalation and ingestion)
Noncancer effects	Ingestion HQ	Ingestion pathway noncancer risk characterization from exposure to all ingestion pathway components for a single chemical
	Inhalation HQ	Inhalation pathway noncancer risk characterization for a single chemical
Copper and lead	Ingestion HQ based on drinking water action level	Ingestion pathway noncancer risk characterization based on groundwater concentration

3.6 Groundwater to Surface Water Pathway Screening Analysis

To characterize potential impacts to ecological receptors, a screening analysis was conducted to determine whether ecological receptors could be affected via the groundwater to surface water pathway. Ecological exposures can occur in surface water when contaminated groundwater intersects a stream. The scenario for the groundwater to surface water pathway is

further discussed in Section 4.3.1.2. The modeling used to estimate the concentrations in surface water due to infiltration of groundwater is described in Section 5.2.2.3.

The screening was conducted using ambient water quality criteria upon which allowable mass loading limits were based. Three chemicals in the analysis were found to have ambient water quality criteria for the protection of aquatic biota: copper, zinc, and lead. The ambient water quality criteria were the criterion “continuous concentrations.” Initial screening calculations for lead in surface water due to infiltration of groundwater indicated that the limiting pathway for lead was based on the drinking water MCL for groundwater. Thus, further analyses on the groundwater to surface water pathway, including the results presented in the risk assessment, only include copper and zinc.

The allowable mass load calculations for the groundwater to surface water pathway are conducted in the same way allowable mass loads are calculated for human exposure to groundwater. The only differences are the way that concentrations are calculated for the surface water body and the use of ambient water quality criteria rather than MCLs or human health benchmarks.

3.7 Probabilistic Modeling Approach

A Monte Carlo simulation was designed to calculate the 90th percentile allowable mass loading for each chemical COC. In a Monte Carlo simulation, the models are run a number of times, with each realization producing a single result (e.g., a single estimate of cancer risk). For this assessment, the Monte Carlo simulation included 10,000 iterations for each scenario (defined as the combination of site/source data, chemical constituent, and liner type). The output of each simulation is, therefore, a distribution of 10,000 values representing the distribution of possible outcomes given the underlying variability and uncertainty in the data used in the analysis. Because the purpose of this assessment was to calculate mass loadings that are protective of human health and the environment, specific waste concentrations were not initially used in the source models. Instead, each simulation was executed using the same unit mass load (e.g., 1 kg/yr).

The input data for the simulation were designed such that results across liner types would be directly comparable. As depicted in Figure 3-2, the data sampled for 10,000 iterations on each combination of WMU and chemical was identical across liner type. In other words, for each WMU and chemical combination, the same site and source data were used for each of the liner types. To ensure comparability of the results across liner type, a source datafile was constructed for each WMU that contained all of the site and source data required for the fate and transport models. Thus, when a simulation was initiated for a given chemical and WMU, the same source data were called regardless of the liner type.

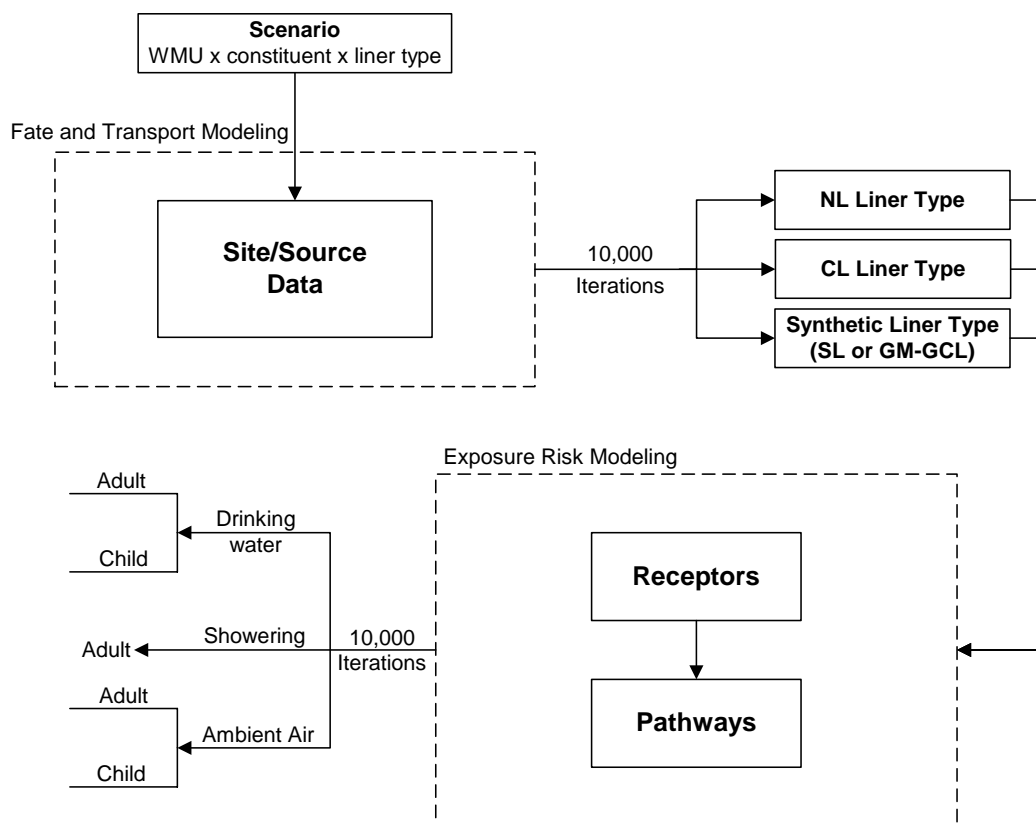


Figure 3-2. Monte Carlo framework.

Each Monte Carlo iteration of the simulation consisted of three major steps:

1. Selecting characteristics of the WMU
2. Selecting characteristics of the environmental setting
3. Selecting exposure parameters for each receptor.

WMUs were randomly selected from the landfill, tanks, and surface impoundments databases, respectively. WMU locations were correlated to one of the EPA Composite Model for Leachate Migration with Transformation Products (EPACMTP) Climate Centers across the contiguous 48 states to represent the national character of this assessment. Exposure parameters were selected randomly based on statistical distributions. For metals, distribution coefficients were also varied based on empirical data. Details on these parameter distributions are presented in Appendices D, E, and H of this document.

3.8 Calculating Allowable Mass Loadings

The allowable mass loadings reflect an annual quantity of a constituent that can be placed in a landfill, surface impoundment, or tank and be protective of human health and the environment. These mass loading limits are defined as mass loadings that result in cancer risks no greater than 1 in one hundred thousand (1E-5) and noncancer HQs no greater than 1 at the 90th percentile of the distribution of risks. Thus, when the mass loading limit for a constituent is

modeled for a distribution of WMUs, the resulting risk distribution will show that the 90th percentile cancer risk is 1E-05 or the 90th percentile HQ is 1.

As noted in Section 3.6, each simulation in the probabilistic modeling was executed using the same unit mass load into the WMU. The modeling is used to obtain a distribution of risk based on the unit mass load. The ratio of the 90th percentile risk from the distribution to the target risk level results in a scaling factor that can be used to adjust the unit mass load to result in the allowable mass load for a constituent. This method for determining the allowable mass loading is appropriate for organic compounds because the model used to go from source to risk is linear. In other words, if the mass load of the chemical in the WMU is doubled, the resulting risk is also doubled. However, the model is not linear for metals, so the same approach cannot be used. For metals, a doubling of the mass in the WMU may not result in a direct doubling of the risk. The nonlinear nature of the model for metals occurs because the distribution coefficients used in the groundwater model are based on nonlinear isotherms. The methodology used for metals requires that the groundwater model converge at a target receptor well concentration or target surface water concentration (e.g., based on health risk or ambient water quality criteria) and then report a scaling factor to calculate the allowable mass loading. The methodology for calculating the allowable mass loadings is discussed in detail in Section 9.

4.0 Source Characterization

This risk assessment provides a national characterization of waste management scenarios for wastes generated in the manufacture of dyes and pigments. In these scenarios, the sources are the WMUs in which dye and pigment wastes could be disposed. How these sources are characterized in terms of their physical dimensions, operating parameters, and location is fundamental to the construction of scenarios for modeling. The scenarios that underpin this assessment are based on an understanding of industry operations and waste management practices that has been derived from secondary data sources.

This analysis evaluates risk in a probabilistic manner. The probabilistic analysis is based on a Monte Carlo simulation that produces a distribution of exposures and risks. The foundation for the Monte Carlo simulation is the source data that define the Monte Carlo iterations. Specifically for this analysis, 10,000 iterations were completed to define a distribution of WMU scenarios. Compiling the source data required characterizing the environmental setting in which waste management occurs and characterizing the WMUs in which waste streams are managed. This section discusses the compilation of the source data for the probabilistic analysis.

Section 4.1 presents an overview of the source data development procedure. Section 4.2 summarizes development of the waste management scenarios evaluated in this risk assessment. Section 4.3 presents the methodologies used to characterize the environmental setting, including delineation of the site layout and environmental setting (e.g., meteorology, climate, and aquifers). Section 4.4 describes how the WMUs were characterized in terms of capacities and surface areas.

4.1 Source Data Development Procedure

To capture the national variation in WMU practices for the Monte Carlo analysis, a database of 10,000 different waste management scenarios was created. These 10,000 scenarios provided the source data for the fate and transport modeling. Figure 4-1 presents an overview of the process used to compile the source data needed for source modeling and fate and transport modeling. These source data are organized into source data files. The source data files contain information on locations and WMUs used in the probabilistic analysis. As shown in Figure 4-1, construction of the source data files involved five tasks, some conducted in parallel and some sequentially. The result was three source data files, one each for landfills, surface impoundments, and tanks.

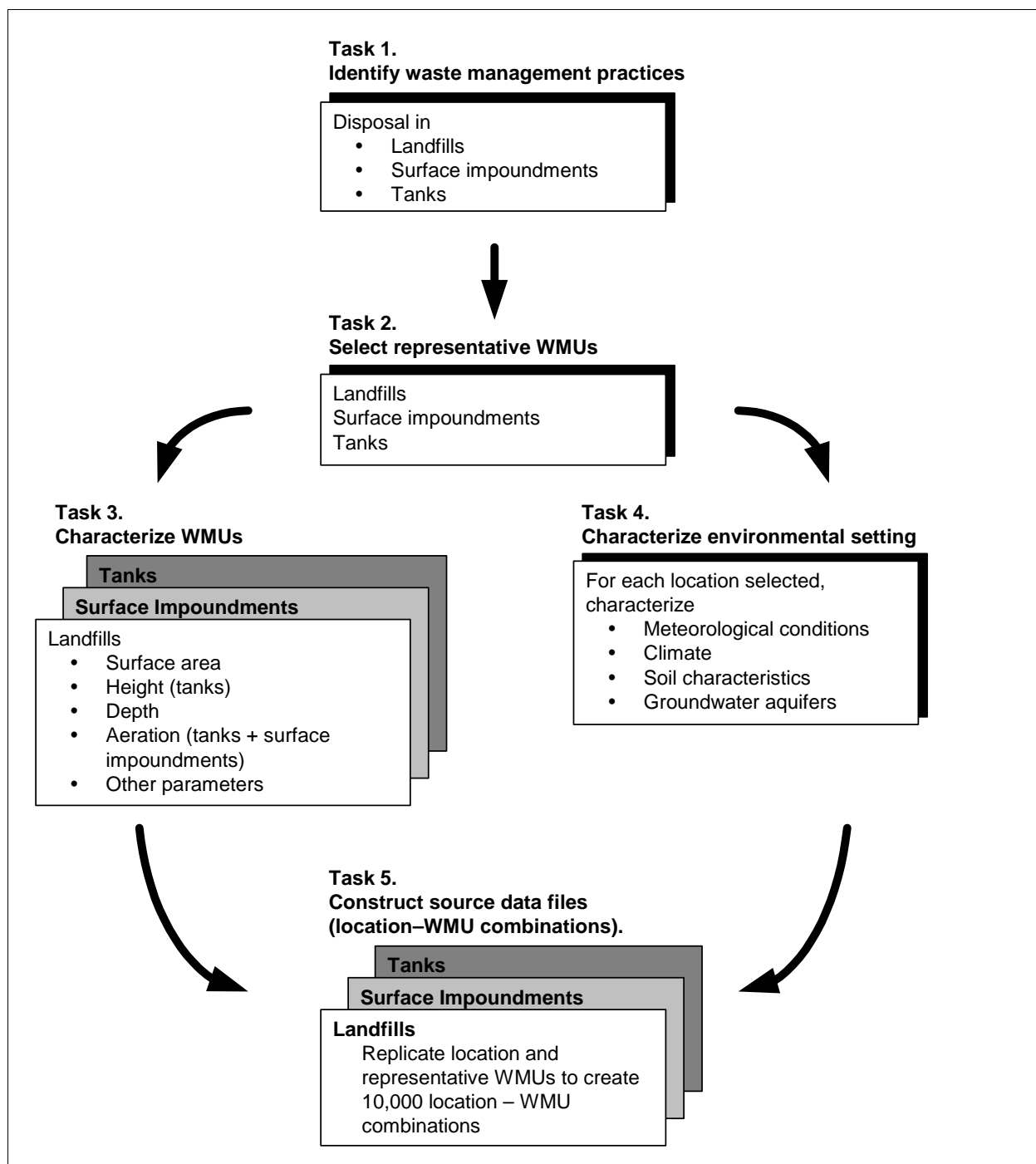


Figure 4-1. Process used to construct source data files used as the basis for the probabilistic analysis.

Task 1. Identify Waste Management Practices

The first task in constructing a database of source characteristics was to identify the waste management practices to be evaluated in this risk assessment. The waste management practices to be evaluated were identified based on data in the open literature, national databases (e.g., Toxics Release Inventory [TRI]), and industry trade associations (see the Listing Background Document for more information). Three WMUs were selected for inclusion in this risk assessment: landfills, surface impoundments, and tanks.

Task 2. Select Representative WMUs

To determine the physical and operating characteristics used in air dispersion modeling and source partition modeling, representative WMUs were selected for each of the three WMU types. First, the databases from which individual WMUs could be selected and characterized were identified, including EPA's Solid Waste (Municipal) Landfill Survey (U.S. EPA, 1988) and the Industrial D database (Schroeder et al., 1987) for landfills; the Surface Impoundment Study (SIS) database (U.S. EPA, 2001a) for surface impoundments; and the Treatment, Storage, and Disposal Facility (TSDF) database (U.S. EPA, 1987) for tanks. Then, representative WMUs were sampled for each iteration of the Monte Carlo simulation from their respective databases.

Task 3. Characterize WMUs

The representative units selected in Task 2 were characterized in this task to develop the physical and operating parameter values (e.g., surface area) that are used in source partition modeling and air dispersion modeling.

Task 4. Characterize Environmental Setting

The location of the WMUs was also based on the survey data discussed in Task 2. These locations were used to characterize meteorology, climate, and aquifers. Meteorological data for a 5-year period were compiled and organized to provide data needed for the air dispersion modeling. Climate data were compiled to provide information used in source modeling and fate and transport modeling (e.g., annual precipitation, temperature). Aquifer types were defined based on the location of the WMU. Both meteorological stations and aquifers were selected to capture the range of conditions found in the contiguous 48 states.

Task 5. Construct Source Data Files (Location-WMU Combinations)

Constructing the 10,000-record source data files for use in the probabilistic analysis involved sampling the representative WMUs selected in Task 2 to produce 10,000 records for each WMU type. The replication of representative WMUs was based on a random sampling of WMUs in each of the databases used in the analysis. Three source data files were generated: one for landfills, one for tanks, and one for surface impoundments. Each record in the source data files was identified by a model run identification (ID) number.

4.2 Waste Management Scenario Development

As discussed in the previous section, the first task in designing a risk assessment for dye and pigment manufacturing waste and waste management practices was to define the waste management scenarios to be evaluated. A waste management scenario consists of a waste stream that is disposed of in a specified type of WMU in a particular location. The waste management scenarios evaluated in this initial risk assessment are summarized here and are described in more detail in Section 3.1.

4.2.1 Characterization of Waste Streams

The dye and pigment wastes were assumed to be mixed with other non-dye and pigment wastes in the disposal unit. Thus, bulk waste parameters (e.g., bulk density, pH, and fraction organic carbon) required to estimate emissions using the source models were parameterized using generic industrial waste characteristics. Whenever possible, distributions were used to characterize the variability in waste parameters. Appendix D lists these parameters, as well as the distributions used.

4.2.2 Waste Management Units

Three types of WMUs were selected for evaluation based on information gathered by EPA. WMUs used for disposal of nonhazardous waste streams by the dye and pigment manufacturing industry include landfills, surface impoundments, and tanks. Because of the large number of manufacturing facilities located across the United States, national databases were used to characterize WMUs accepting dye and pigment waste for disposal. Section 4.3 describes in detail the data sources used for characterizing each type of WMU.

4.2.3 Constituents

In this risk assessment, 35 constituents of concern (COCs) were selected for evaluation—four metal constituents, one inorganic constituent, and 30 organic constituents. Table 3-2 lists these COCs along with their Chemical Abstract Service (CAS) numbers. Appendix B presents physical and chemical properties for each of the constituents evaluated in this risk assessment, and Appendix C presents distribution coefficients (i.e., K_d) for the metals.

4.3 Site Characterization

The site characteristics used in this analysis were based on conceptual site layouts and regional characterization of environmental parameters. The conceptual site layouts define the area in the immediate vicinity of the WMU. They also define the geographic relationship among important features, such as the WMU boundary, resident location, and streams. There are three conceptual site layouts used in this analysis that are evaluated at each of the locations selected for the analysis, all located within the contiguous 48 states. These locations were used to capture national variability in meteorology, climate, and aquifers.

4.3.1 Conceptual Site Layouts

This risk assessment was based on site layouts that are conceptual rather than site-specific. The site layouts were designed to capture possible relationships between a WMU and individual receptors. Geographic features that are important for determining exposures to chemicals released from the WMU (e.g., waterbody) were located relative to the WMU boundary.

Figures 4-2 and 4-3 show the conceptual or general site layouts, including the WMU boundaries, the buffer area (i.e., an area between the WMU and the nearest human receptor), and the resident location. These site layouts were used to model two possible land use scenarios that may exist in areas surrounding the WMUs:

- Residential aboveground scenario (Figure 4-2)
- Residential groundwater scenario (Figure 4-3).

The WMU size and the distance to the resident receptor were both varied as part of the Monte Carlo analysis. In addition, a groundwater-to-surface-water scenario was used to evaluate ecological impacts.

4.3.1.1 WMU Boundaries. The WMU is assumed to be located on the property line of the facility to which it belongs. Adjacent to the WMU is a buffer area within which there is assumed to be no human activity that would result in human exposure. That is, there are no residences within the buffer. The buffer area lies between the WMU boundary and the resident location, agricultural field, or waterbody, depending on the scenario being modeled.

4.3.1.2 Modeling Scenarios. The assessment considered two modeling scenarios based on residential exposure, one for the aboveground pathway and one for the groundwater pathway. Two separate fate and transport scenarios were established for the groundwater and air pathways. The decision to use separate scenarios for the two pathways was based on differences in timeframe and location of exposure. An additional scenario was also considered to assess ecological impacts due to the groundwater to surface water pathway.

Residential Scenarios—Aboveground Pathway. The residential scenario was used to estimate risks to receptors (i.e., adult and child residents) living in the vicinity of the WMUs. Surveys conducted to support the *Hazardous Waste Treatment, Storage, and Disposal Facilities - Organic Air Emissions Standards for Process Vents and Equipment Leaks Final Rule* (55 FR 25454) have shown that the closest residence to a WMU boundary is approximately 75 m, and the median or central tendency distance from the WMU boundary is approximately 300 m. The 75 m distance is taken to be the 10th percentile closest distance. Using these values, a normal distribution of resident locations was developed for the Monte Carlo analysis. It has a median value of 300 m, 75 m for the 10th percentile closest distance, and 525 m for the 90th percentile farthest distance. The distance from the WMU boundary to the resident location was selected from this distribution for each iteration of the Monte Carlo analysis. Values selected were constrained to be between 50 and 550 m so as to avoid extreme values that would be inconsistent with the general scenario described by the site layout.

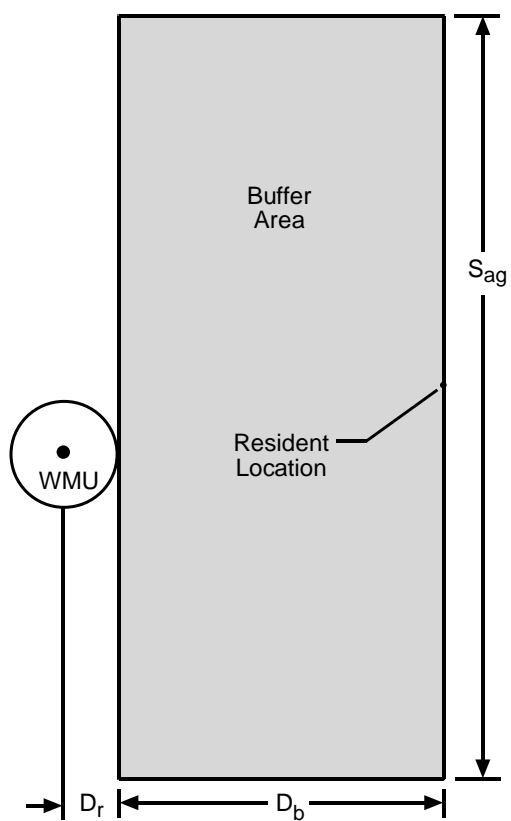


Figure 4-2. Conceptual site layout for residential aboveground scenario.

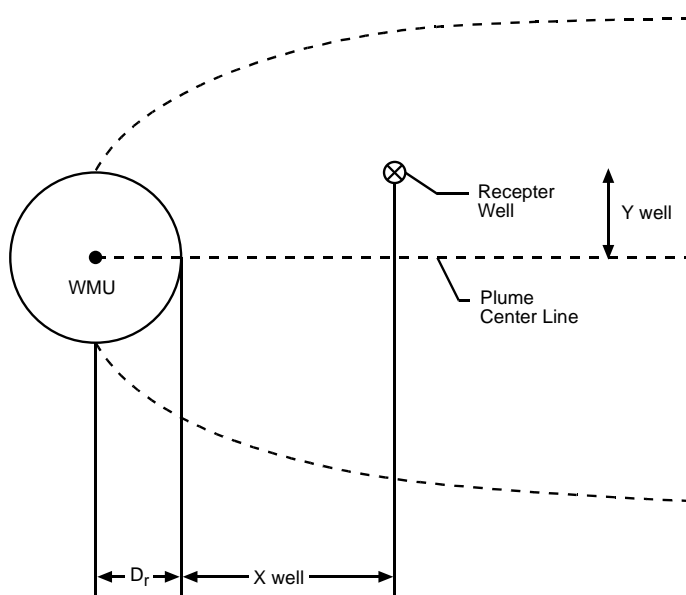


Figure 4-3. Conceptual site layout for residential groundwater scenario.

This site layout must also be oriented in terms of direction. In this assessment, the resident location was selected based on the most affected location for a given distance. In other words, receptors were always located in the predominant wind direction.

Residential Scenarios–Groundwater Scenario. Residential groundwater exposure is calculated based on residential use of well water. The receptor well is placed at a downgradient distance up to 1 mile, based on a nationwide distribution of nearest downgradient residential wells from Subtitle D municipal landfills (U.S. EPA, 1988). This distribution is provided in Appendix E. These data, as well as limits on the lateral direction from the plume centerline and depth below the water table (i.e., z-well) of the well, are discussed in Section 5.2.2.2.

Groundwater-to-Surface-Water Scenario. Ecological exposures occur in surface water contaminated when contaminated groundwater intersects a stream. The stream is assumed to be downgradient of the WMU. The distance to the stream is derived by placing the WMU in an average-sized watershed for a third-order stream and randomly locating the WMU in the watershed. Replicating the random location of the WMU provides a distribution of distance from the WMU to the third-order stream used for the groundwater-to-surface-water scenario. A third-order stream refers to a type of stream segment classification. In this classification scheme, a first-order stream segment is one with no tributaries. That is, a first-order stream segment receives all of its flow from runoff from the surrounding watershed soils. A second-order stream segment is produced when two first-order stream segments come together. A third-order stream segment occurs when two second-order segments come together. (A third-order stream segment is not produced when a second-order and a first-order stream segment combine.) The third-order stream segment, therefore, has the combined flow of at least two second-order stream segments. The third-order stream was selected because it reasonably represents the smallest waterbody that can support a multicompartiment ecological food web and would routinely support recreational fishing of consumable fish.

4.3.2 Regional Environmental Setting

The purpose of the dyes and pigments listing risk assessment was to develop national distributions of allowable mass loadings that would be protective of human health and the environment. The assessment was conducted using a fixed conceptual site model that could exist anywhere in the contiguous 48 states. Other parameters that would affect the results of this risk assessment are those that reflect regional environmental conditions (e.g., meteorology and groundwater hydrology), differences in WMU design, and differences in waste stream characteristics. In order to define the environmental setting, a specific location needed to be assigned for each iteration of the model. The surveys used to define the characteristics of WMUs also provided specific locations for each unit in the survey. Each time a specific unit was chosen for an iteration of the Monte Carlo assessment, the location of the unit was also selected and remained correlated to that specific unit. The following sections describe how meteorological and hydrogeologic parameters were mapped to WMU locations.

4.3.2.1 Meteorological Station Locations. To identify meteorological stations, locations of WMUs were obtained from survey data (see subsequent sections for location determination). These WMU locations were then cross-referenced to EPACMTP climate regions that were in turn cross-referenced to one of 57 meteorological regions. Each

meteorological region is represented by one meteorological station. Meteorological region boundaries and representative meteorological station were developed for the Industrial Waste Air (IWAIR) model (U.S. EPA, 2002a). Development of the meteorological regions assessed the factors most important for the inhalation pathway risk modeling and involved two considerations:

1. Identification of contiguous areas that are sufficiently similar with regard to the parameters that affect dispersion that they can reasonably be represented by one meteorological station. The parameters used for this determination were surface-level meteorological data (e.g., wind patterns and atmospheric stability); physiographic features (e.g., mountains, plains); Bailey's ecoregions and subregions; and land cover (e.g., forest, urban areas).
2. Selection of one meteorological station to represent each contiguous area. Station selection considered the following parameters: industrial activity, population density, location within the area, years of meteorological data available, and average windspeed.

Table 4-1 lists the 57 stations chosen; Figure 4-4 shows the selected stations and their assigned regions for the contiguous 48 states. The IWAIR Technical Background Document (U.S. EPA, 2002a) describes the selection process in detail.

Table 4-1. Surface-Level Meteorological Stations in IWAIR, by State

Station Number	Station Name	State
13963	Little Rock/Adams Field	AR
23183	Phoenix/Sky Harbor International Airport	AZ
93193	Fresno/Air Terminal	CA
23174	Los Angeles/International Airport	CA
24257	Redding/AAF	CA
23234	San Francisco/International Airport	CA
23062	Denver/Stapleton International Airport	CO
14740	Hartford/Bradley International Airport	CT
12839	Miami/International Airport	FL
12842	Tampa/International Airport	FL
13874	Atlanta/Atlanta-Hartsfield International	GA
03813	Macon/Lewis B Wilson Airport	GA
94910	Waterloo/Municipal Airport	IA
24131	Boise/Air Terminal	ID
94846	Chicago/O'Hare International Airport	IL

(continued)

Table 4-1. (continued)

Station Number	Station Name	State
03937	Lake Charles/Municipal Airport	LA
12916	New Orleans/International Airport	LA
13957	Shreveport/Regional Airport	LA
14764	Portland/International Jetport	ME
94847	Detroit/Metropolitan Airport	MI
14840	Muskegon/County Airport	MI
14922	Minneapolis-St Paul/International Airport	MN
13994	St. Louis/Lambert International Airport	MO
13865	Meridian/Key Field	MS
24033	Billings/Logan International Airport	MT
03812	Asheville/Regional Airport	NC
13722	Raleigh/Raleigh-Durham Airport	NC
24011	Bismarck/Municipal Airport	ND
14935	Grand Island/Airport	NE
23050	Albuquerque/International Airport	NM
23169	Las Vegas/McCarran International Airport	NV
24128	Winnemucca/WSO Airport	NV
14820	Cleveland/Hopkins International Airport	OH
93815	Dayton/International Airport	OH
13968	Tulsa/International Airport	OK
94224	Astoria/Clatsop County Airport	OR
24232	Salem/McNary Field	OR
14751	Harrisburg/Capital City Airport	PA
13739	Philadelphia/International Airport	PA
14778	Williamsport-Lycoming/County	PA
11641	San Juan/Isla Verde International Airport	PR
13880	Charleston/International Airport	SC
13877	Bristol/Tri City Airport	TN
13897	Nashville/Metro Airport	TN
23047	Amarillo/International Airport	TX
13958	Austin/Municipal Airport	TX
12924	Corpus Christi/International Airport	TX
03927	Dallas/Fort Worth/Regional Airport	TX

(continued)

Table 4-1. (continued)

Station Number	Station Name	State
12960	Houston/Intercontinental Airport	TX
23023	Midland/Regional Air Terminal	TX
24127	Salt Lake City/International Airport	UT
13737	Norfolk/International Airport	VA
14742	Burlington/International Airport	VT
24233	Seattle/Seattle-Tacoma International	WA
24157	Spokane/International Airport	WA
03860	Huntington/Tri-State Airport	WV
24089	Casper/Natrona Co International Airport	WY

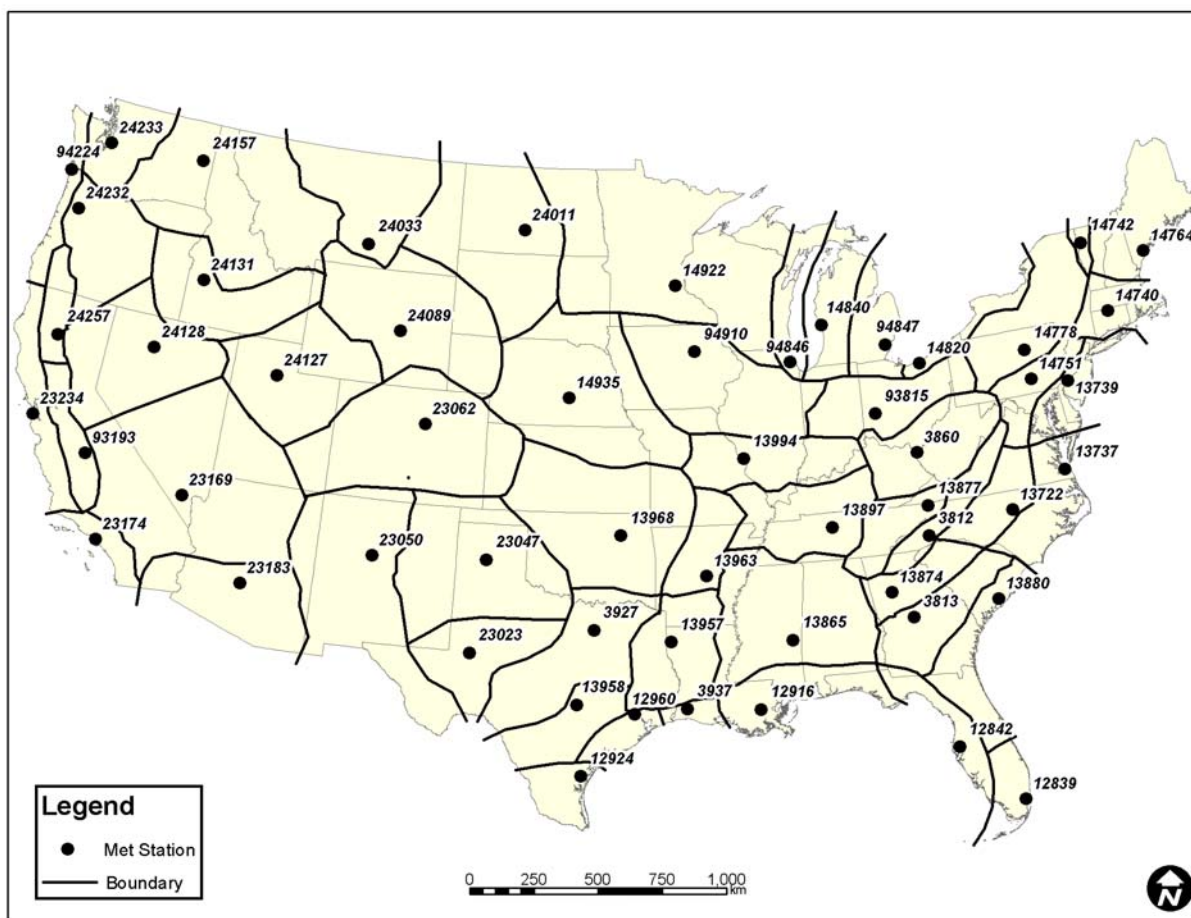


Figure 4-4. Meteorological stations and region boundaries for the contiguous 48 states.

4.3.2.2 Hydrogeologic Environments and Aquifer Properties. Locations evaluated in this risk assessment were established by the selection of WMUs and are used to define a regional framework for the collection of aquifer data. For aquifer properties (used by the source partition and groundwater models), hydrogeologic environments had to be designated for each of the locations modeled so that correlated, national aquifer property data from the American Petroleum Institute (API) Hydrogeologic Database (HGDB; Newell et al., 1989; Newell et al., 1990) could be used in the analysis. The groundwater model, the EPACMTP, uses the HGDB data to specify probability distributions for each of four hydrogeologic parameters:

- Unsaturated zone thickness
- Aquifer thickness
- Hydraulic gradient
- Longitudinal hydraulic conductivity.

Average aquifer/vadose zone temperature was also required for the groundwater model. These data were obtained from a map of groundwater temperatures for the contiguous 48 states in the *Water Encyclopedia* (van der Leeden et al., 1990). The remaining parameters were developed as described below.

The HGDB provides correlated data on the four hydrogeologic parameters listed above and an aquifer classification based on approximately 400 hazardous waste sites nationwide, grouped according to 12 hydrogeologic environments described in Newell et al. (1990) and shown in Table 4-2. The *EPACMTP Parameter and Data Document* (U.S. EPA, 2003b) provides

Table 4-2. Twelve Hydrogeologic Environments in EPACMTP

Code	Description
01	Metamorphic and igneous
02	Bedded sedimentary rock
03	Till over sedimentary rock
04	Sand and gravel
05	Alluvial basins valleys and fans
06	River valleys and flood plains with overbank deposits
07	River valleys and flood plains without overbank deposits
08	Outwash
09	Till and till over outwash
10	Unconsolidated and semiconsolidated shallow aquifers
11	Coastal beaches
12	Solution limestone

Source: Newell et al. (1990).

empirical distributions of values for each of the four hydrogeologic parameters for each of the hydrogeologic environments based on the HGDB.¹ These data are also presented in Appendix E.

As noted above, each the of WMU locations used in this analysis were also assigned to hydrogeologic environments. Thus, the specific empirical distribution used for a given iteration of the model was determined by the assignment of the hydrogeologic environment. Once the hydrogeologic environments were assigned, a preprocessing run of EPACMTP was conducted to construct a set of randomly generated but correlated hydrogeologic parameter values for each occurrence of the hydrogeologic environments in the 10,000-record location data set. Missing values in the HGDB data set were filled using correlations, as described in U.S. EPA (2003b). The unsaturated zone thickness generated from the preprocessing run was also an input parameter required in the surface impoundment source model. The output of the source modeling, along with the hydrogeologic parameter input values used, was reported to an output file for use as input to the subsequent groundwater modeling analysis. Thus, the inputs to the surface impoundment and the groundwater model were synchronized to ensure consistency.

4.4 Characterization of Waste Management Units

This section characterizes specific WMUs with respect to capacity and dimensions (e.g., area, depth). These dimensions and operating characteristics are important determinants of the modeled emission rates, dispersion factors, and leachate concentrations used to estimate exposures. Source parameters that are specifically used to estimate emissions are discussed in Section 5.1, Source Modeling of Constituent Releases.

4.4.1 Landfills

To model risks associated with disposal of dyes and pigments industry wastes in landfills, detailed information was compiled about landfill characteristics. This section discusses the methods and data used to characterize the landfills in which dye and pigment wastes may potentially be disposed. These data are then used as inputs to the source partition model, the air dispersion model, and the groundwater fate and transport model (EPACMTP).

The main sources of data used to characterize landfills for the current modeling analysis are introduced in Section 4.4.1.1. Summarized in Section 4.4.1.2 are the procedures that were used to create the Hazardous Waste Identification Rule (HWIR) 1995 landfill database from the results of the Industrial D Screening Survey; that is, screening the reported data; generating random, correlated values for reported survey values that were missing or deemed unreliable; and then assigning an aquifer type, climate region, and groundwater temperature to each landfill site in the database. The distributions of municipal landfill areas are the only data used for this modeling analysis from the Municipal Landfill Survey and are presented in Section 4.4.1.3. The process by which these data were segregated and recombined to create the database of landfill sites used for the current modeling analysis is summarized in Section 4.4.1.4. The infiltration and recharge rates used for each liner scenario and the assumptions embedded in the conceptual

¹ EPACMTP also includes a thirteenth environment, with national average properties, for sites that cannot be easily classified into the 12 HGDB hydrogeologic environments.

model of the landfill in which dye and pigment wastes are disposed are presented in Sections 4.4.1.5, and 4.4.1.6, respectively.

4.4.1.1 Data Sources. The landfill scenario evaluated in this analysis was the disposal of dye and pigment wastes in municipal landfills. EPA's Solid Waste (Municipal) Landfill Survey (U.S. EPA, 1988) was used to define the area of the landfills. This survey did not provide all of the parameters required for modeling emissions and leachate concentrations from landfills. The groundwater modeling database of landfill sites created for the 1995 proposed HWIR (U.S. EPA, 1997b) was used to further define landfill characteristics. The HWIR 1995 database of landfill sites is based on the results of the 1985 Screening Survey of Industrial Subtitle D Establishments, referred to as the Industrial D Screening Survey (Schroeder et al., 1987). The final database used in the modeling analysis for the dyes and pigments industry listing determination was created through manipulation of the HWIR 1995 landfill site database to more accurately reflect the waste management practices used by the dyes and pigments manufacturing industry, as explained below in Section 4.4.1.4.

4.4.1.2 Summary of the HWIR 1995 Landfill Database Creation. The Industrial D Screening Survey was designed to collect information about nonhazardous (Resource Conservation and Recovery Act [RCRA] Subtitle D) waste management practices at industrial landfills, wastepiles, land application units, and surface impoundments across the United States. This survey collected information on land-based Industrial D waste management operations for 17 industry groups.² Data from this survey have been used to represent Industrial D WMU characteristics in a variety of RCRA regulatory initiatives. Although the Industrial D data are more than 10 years old, they are the largest consistent set of data available on Industrial D WMU dimensions and characteristics. Information on the survey design, response rates, and overall data quality and completeness is provided in Schroeder et al. (1987), Clickner (1988), and Clickner and Craig (1988).

Landfill Characterization. The 1986 Industrial D Screening Survey (Schroeder et al., 1987) provided landfill data consisting of 824 observations of landfill facility locations, area, number of units in the facility, facility design capacity, total remaining facility capacity, and the relative weight of each facility. The relative weight was assigned based on the total number of employees working at the facility, and it reflects the quantity of the waste managed in that facility. When more than one unit was present at a facility, average unit values were calculated for area and capacity by dividing the reported data value by the number of units at the site. Of the original 824 landfills in the Industrial D Screening Survey, the following were deleted during creation of the HWIR 1995 landfill site database:

- 21 landfills were deleted because facility location was missing
- 9 landfills were deleted because landfill area was missing

² Industry groups are as follows: (1) organic chemicals; (2) primary iron and steel; (3) fertilizer and agricultural chemicals; (4) electric power generation; (5) plastic and resins; (6) inorganic chemicals; (7) stone, clay, glass, and concrete; (8) pulp and paper; (9) primary nonferrous metals; (10) food and kindred products; (11) water treatment; (12) petroleum refining; (13) rubber and miscellaneous products; (14) transportation equipment; (15) selected chemical and allied products; (16) textiles; and (17) leather and leather products.

- 2 landfills in Hawaii were deleted because they were outside of the contiguous 48 states
- 2 landfills were deleted because the area and total capacity data were deemed to be highly unreliable.

In this way, the set of 790 landfills in the HWIR 1995 database was established; the geographic locations of these landfills are presented in Figure 4-5. The data representing these 790 landfills were then screened by placing two constraints on the total landfill design capacity to eliminate unrealistic observations, as described in U.S. EPA (2003b). First, total landfill capacity was constrained to be greater than the remaining capacity; and second, the landfill depth (calculated by dividing the unit's total capacity by its area) was constrained to be greater than or equal to 2 ft (0.67 m) and less than or equal to 33 ft (10m). The depth constraints presented in U.S. EPA (2003b) were adapted from the previous Toxicity Characteristic rule effort. The result was that 232 records had a reported total landfill capacity that violated at least one of these criteria; additionally, 91 facilities were missing data on total capacity. Thus, total landfill capacity was missing or screened for 323 landfills.



Figure 4-5. Geographic locations of landfill waste WMUs.

In cases where the landfill depth or remaining capacity constraints were violated and in cases where the landfill capacity value was missing, the missing/screened capacity value was automatically replaced with a randomly generated, correlated value by the EPACMTP model during run time. This was accomplished by generating a random realization from the volume probability distribution conditioned on area, assuming that the unit area value was more likely to be correctly reported, as described in U.S. EPA (2003b). The joint distribution was assumed to be lognormal and was derived from the non-missing unit area/volume pairs that met the unit depth and remaining capacity constraints. The EPACMTP model first generates a missing

landfill capacity value from the conditional distribution and then calculates the landfill depth value (the required model input value) by dividing the unit volume by the area.

EPACMTP is EPA's probabilistic groundwater fate and transport model (U.S. EPA, 1997a,b; 2003a,b). The regional, site-based modeling procedure employed by EPACMTP is based on data from the Industrial D Screening Survey, as described above. Because the Industrial D Screening Survey includes only facility-specific data, dynamic linkages to other data sources are used to generate the additional, correlated input parameters required to perform the groundwater fate and transport modeling for each site. That is, for use in EPACMTP modeling analyses performed for HWIR 1995, each of the 790 landfill sites in the Subtitle D survey database was classified according to the type of aquifer underlying the site and the closest climate center in order to provide links to the databases of hydrogeologic data (HGDB) and infiltration/recharge data (based on Hydrologic Evaluation of Landfill Performance [HELP] water-balance modeling), as summarized below. Additional details can be found in the *EPACMTP Technical Background Document* and the *EPACMTP Parameters and Data Document* (U.S. EPA, 2003a,b).

Subsurface Characterization. The HGDB (Newell et al., 1989; U.S. EPA, 1997b) contains site-specific data on groundwater parameters (aquifer thickness, depth to groundwater, hydraulic gradient, and hydraulic conductivity) collected by independent investigators at approximately 400 hazardous waste sites throughout the United States. These site-specific data were regrouped into 12 hydrogeologic environments, based on the United States Geological Survey (USGS) classification of aquifer regions (Heath, 1984). The result is a database of aquifer types, with each aquifer type consisting of an empirical distribution of values for each of the four aquifer parameters. The geographic location of the site was used in combination with USGS state-by-state aquifer maps (Heath, 1984) to assign each of the 790 landfill sites in the database to one of the 12 hydrogeologic environments. Sites that could not be classified into one of the 12 categories were assigned as "other," i.e., they were assigned to environment number 13. Environment 13 represents the average of all sample values for each of the four parameters. The HGDB data are included in one of the EPACMTP input data files; at run time, for each site selected from the landfill database, the model determines a correlated set of values for the hydrogeological parameters from the HGDB data set for the hydrogeologic environment assigned to that landfill site.

Climate Characterization. Infiltration and recharge rates for use in EPACMTP modeling applications have been estimated for selected soil types at cities around the country through the use of the HELP water-balance model (U.S. EPA, 1994a, 1994b, 1997b, 2002b, 2003a). Using the Soil Conservation Service's (SCS's) county-by-county soil mapping database, three soil textures were defined: coarse-, medium-, and fine-grained soils. Using National Oceanic and Atmospheric Administration (NOAA) data on precipitation and evaporation rates in the United States, 102 cities were selected as climatic centers for the HELP model. For each selected city, historical climatic data were used to develop an ambient regional recharge rate as a function of site location and soil type; likewise, infiltration rates were developed for each type of WMU, as a function of site location, liner type (if any), and soil type. The geographic location of the site was used to assign each of the 790 landfill sites in the database to one of the 102 climate centers. In a procedure similar to that used for the hydrogeologic data, the infiltration and recharge data are included in one of the EPACMTP input

data files; at run time, for each site selected from the landfill database, the model determines a correlated pair of infiltration and recharge rates for the climate center assigned to that landfill site.

Additionally, because the groundwater temperature at each landfill site is a required input to the groundwater fate and transport modeling but was not included in the Industrial D Screening Survey, these data were added during development of the HWIR 1995 landfill site database. Information on average annual temperatures in shallow groundwater systems from Todd (1980) was used to assign a temperature value to each landfill in the HWIR 1995 database, based on the unit's geographical location. For each site, the assigned temperature was an average of the upper and lower values for that temperature region, as shown in Figure 4-6. In other words, all landfills located in the band between 10° and 15° were assigned a temperature value of 12.5° C.

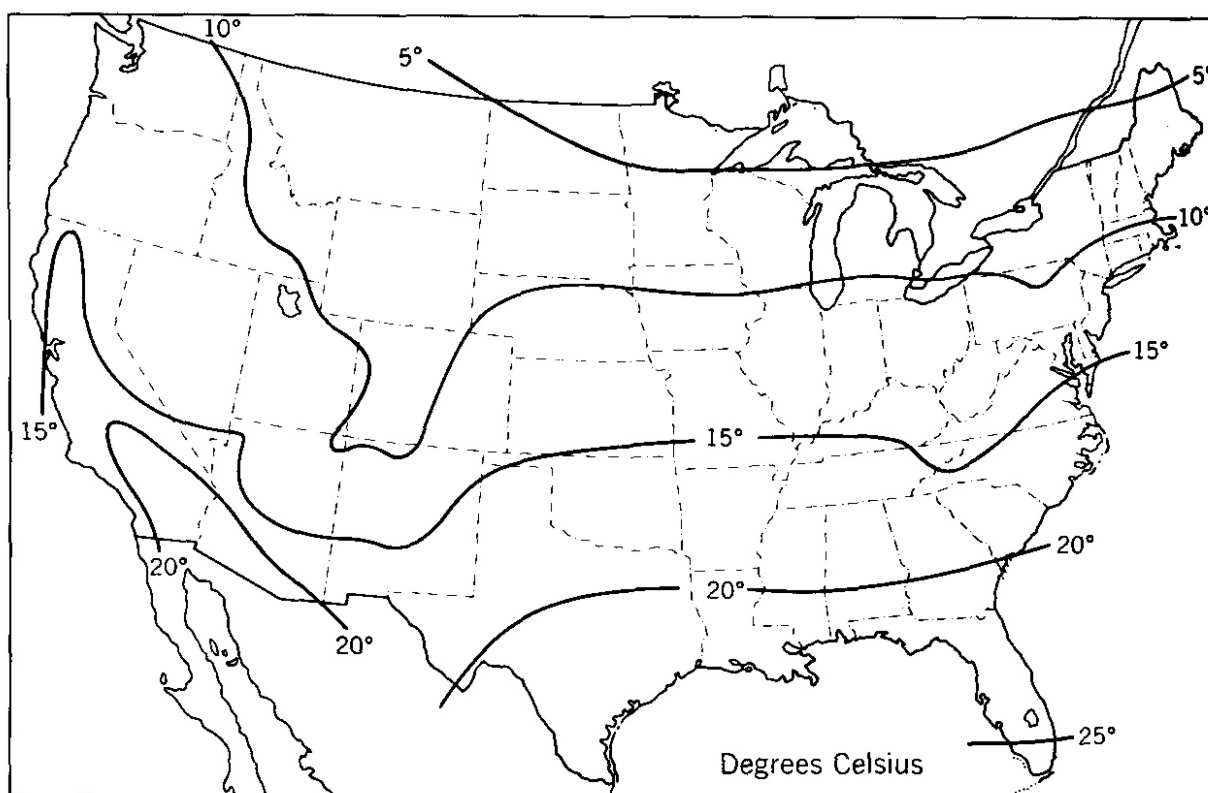


Figure 4-6. Groundwater temperature distribution for shallow aquifers in the United States (based on Todd, 1980).

4.4.1.3 Municipal Landfill Area Distribution. An empirical distribution of landfill areas derived from EPA's Solid Waste (Municipal) Landfill Survey (U.S. EPA, 1988) was used to generate the areas for municipal landfills, as explained in Section 4.4.1.4. This empirical distribution of municipal landfill areas is presented in Table 4-3.

Table 4-3. Empirical Distribution of Municipal Landfill Areas (from U.S. EPA, 1988)

Area Range (m ²)	Relative Probability
4,000 to 8,090	0.10
8,090 to 20,200	0.15
20,200 to 60,700	0.25
60,700 to 194,000	0.25
194,000 to 420,000	0.15
420,000 to 9,350,000	0.10
Total	1.00

4.4.1.4 Summary of Landfill Database Creation for the Current Modeling Analysis.

Based on available data, it is expected that wastes generated from the manufacture of organic dyes and pigments will be primarily disposed of in municipal landfills. However, the existing database of landfill sites used by EPACMTP (U.S. EPA, 1997a,b; 2003a,b) (described above) comprises only industrial facilities and is heavily weighted toward onsite units. For these reasons, EPA decided to use available data to create a new database of landfill characteristics that more accurately reflect the waste management practices in the dye and pigment industry.

As described in Section 4.4.1.2 above, the HWIR 1995 landfill site database that is typically used for EPACMTP modeling (U.S. EPA, 1997a,b; 2003a,b) was generated from the *1985 Screening Survey of Industrial Subtitle D Establishments* (Schroeder et al., 1987). This database consists of 790 industrial landfill units (WMU area and depth) located across the contiguous 48 states with regionally correlated climate and hydrogeological data. The steps in the processing of these data were as follows:

1. All sites in the landfill database were segregated into one set of onsite units (724 sites, approximately 92%) and another set of offsite units (66 sites, approximately 8%) (using Table E-1 in Appendix E of U.S. EPA, 2001b). The EPACMTP model was then executed using only the data for the offsite units, generating a complete input data set consisting of 10,000 records.
2. The characteristics of municipal landfills are expected to be similar to those of offsite industrial landfills, with the exception of area and depth. An empirical distribution of landfill areas derived from EPA's Solid Waste (Municipal) Landfill Survey (U.S. EPA, 1988) was used to generate area data (see Table 4-3, above), but no data on the depth of municipal landfills was available. Consequently, the empirical distribution of municipal landfill areas was randomly sampled to generate set of 10,000 values.
3. Municipal landfill depths were then generated using area and capacity correlations extracted from the Industrial D database (U.S. EPA, 1997b).

4. The 10,000 data pairs of municipal landfill areas and derived depths were combined with a duplicate of the offsite industrial data, replacing the offsite areas and depths. The result was a complete data set consisting of 10,000 records.

4.4.1.5 Infiltration and Recharge Data Used for Each Landfill Liner Scenario. The modeling conducted for the municipal landfill scenario included analyses for each of three liner scenarios: no liner, clay liner, and composite liner. For the no-liner and clay-liner scenarios, the default EPACMTP databases of landfill infiltration rates and ambient regional recharge rates (generated by the HELP model) were used (U.S. EPA, 1994a, 1994b, 2002b, 2003a). For the composite liner type, two scenarios, each including empirical distributions of infiltration rates, were compiled from the measured monthly average leak detection system (LDS) flow rates for clay composite-lined landfill cells reported by TetraTech (2001). The infiltration data in this report cover four general liner configurations:

1. GM: High-density polyethylene (HDPE) geomembrane over native soil
2. GM/CCL: 1 to 1.5 mm thick geomembrane overlying a 3 ft thick compacted clay liner
3. GM/GCL: 1 to 1.5 mm thick geomembrane overlying a 6 mm thick geosynthetic clay liner
4. GM/GCL/GM: A 6 mm thick geosynthetic clay liner between two 1 to 1.5 mm thick geomembranes
5. GM/GCL/CCL: 1 to 1.5 mm thick geomembrane overlying a 6 mm thick geosynthetic clay liner and a 3 ft thick compacted clay liner.

The first scenario, referred to as the GM/GCL scenario, includes a distribution of composite liner infiltration rates only for the GM/GCL configuration. The second scenario, referred to as the synthetic liner scenario, includes a distribution that incorporates four of the above composite liner configurations: GM/CCL, GM/GCL, GM/GCL/GM, and GM/GCL/CCL.

Although there is the potential for water to be released during the consolidation of the compacted clay liner, yielding an unknown contribution of water to the LDS flow rate, EPA expects the problem to be much less significant at the higher infiltration rates that will drive the higher end of the resulting risk distribution. For this reason, the GM/CCL data points were retained in the synthetic liner scenario. However, because EPA wanted to only consider composite lined landfills in accordance with RCRA, the GM data points were removed from both scenarios. Removing these data leaves an infiltration rate distribution of 53 data points. These data were then further screened against the following criteria:

- Only infiltration rates representing a monitoring duration of longer than 10 months were included in the final data set in an attempt to ensure that the values represented true long-term average flow rates.

- Only infiltration rates for municipal solid waste units were included in the final data set; that is, data for units containing municipal waste mixed with ash were excluded.
- Only infiltration rates for units constructed following a construction quality assurance plan were included in the final data set.

After applying the above criteria, the final data set consisted of 40 data points. Of these 40, 22 corresponded to the GM/GCL configuration. These two sets of data were used to define the two composite liner distributions for the risk assessment. Summary statistics on both distributions are presented in Table 4-4 below.

**Table 4-4. Distributions of Composite Liner Infiltration Rates
(subsets of data reported in TetraTech, 2001)**

Percentile	Infiltration Rate (m/yr)	
	Synthetic Liner Scenario n = 40	GM/GCL Scenario n = 22
0	0.00E+00	0.00E+00
10	0.00E+00	0.00E+00
20	0.00E+00	0.00E+00
30	0.00E+00	0.00E+00
40	7.31E-05	0.00E+00
50	1.28E-04	0.00E+00
60	2.19E-04	2.19E-05
70	1.30E-03	7.30E-05
80	3.31E-03	7.30E-05
90	5.63E-03	1.730E-04
100	1.17E-02	4.015E-04

4.4.1.6 Landfill Conceptual Model. Landfill data collection assumed that only one type of landfill is used for disposal of waste (i.e., that there are no significant differences in the design of landfills depending on size or purpose, other than the presence or absence of an engineered liner). Other significant assumptions were that the landfill is excavated below ground surface, the unit receives waste for 30 years, and that the landfill is finally capped with soil cover to establish a vegetative cover after a cell is filled. The assumption of a soil cover is a conservative one because many landfills are now being closed with synthetic covers below a soil and vegetation cover. It was also assumed that there are controls in place to prevent overland transport of constituents to adjacent land areas by runoff or erosion.

4.4.2 Surface Impoundments

A surface impoundment is an excavation or diked area typically used for the treatment, storage, or disposal of liquids or sludges containing free liquids. Liquids and solids typically separate in a surface impoundment by gravity settling. Liquids from surface impoundments are removed by draining, evaporation, or flow through an outlet structure. Accumulated solids are typically removed by dredging during impoundment operation or at the time of closure. The EPA estimates that, in the 1990s, there were approximately 18,000 industrial surface impoundments in use throughout the United States (U.S. EPA, 2001a). They are used for waste management by a number of industries, including chemical manufacturers, food processors, oil refineries, primary and fabricated metals, paper plants, and other commercial facilities. See the listing background document for more information about the use of surface impoundments in the dye and pigment industry.

To model risks associated with the management of dye and pigment wastes in surface impoundments, detailed information was compiled about surface impoundment characteristics. This section discusses the methods and data used to sample and characterize surface impoundments. These data are then used as inputs to the source partition model, the air dispersion model, and the groundwater fate and transport model (EPACMTP).

The primary data source used to characterize surface impoundments for the current modeling analysis, the EPA's Surface Impoundment Study (SIS), is presented in Section 4.4.2.1. Section 4.4.2.2 summarizes the procedures that were used to create the surface impoundment database from the results of the SIS, including default values used to replace missing data and the assignment of aquifer type, climate region, and groundwater temperature to each surface impoundment site in the database. The infiltration and recharge rates used for each liner scenario and the assumptions embedded in the conceptual model of the surface impoundments used in dye and pigment waste management are presented in Sections 4.4.2.3 and 4.4.2.4, respectively.

4.4.2.1 Data Sources. Although surface impoundments were included in the 1985 Industrial D Screening Survey, EPA has adopted the results of the more recent SIS as the data source for the nationwide database of surface impoundment sites used in the EPA's environmental modeling. The SIS was a 5-year study of nonhazardous (Subtitle D) industrial surface impoundments in the United States (U.S. EPA, 2001a). As part of the effort to develop the Industrial Waste Evaluation Model (IWEM) (a simplified version of the EPACMTP model tailored for a specific application) in support of the *Guide for Industrial Waste Management* (U.S. EPA, 2003c), the SIS results were screened and compiled into a surface impoundment site database for use with EPACMTP.

4.4.2.2 Summary of the IWEM Surface Impoundment Database Creation. For the IWEM modeling effort, data from the SIS were used to create a new database of impoundment characteristics comprising 494 surface impoundment units located at 144 facilities throughout the contiguous United States. The SIS data used to create the IWEM surface impoundment site database include impoundment location, area, operating depth (depth of ponding in the impoundment), operational life of the impoundment, depth of the surface impoundment base below the ground surface, predominant soil type in the unsaturated zone beneath the

impoundment, proximity of the impoundment to the closest surface waterbody, and the relative weight of each unit. The relative weight of each impoundment in the database reflects the quantity of the waste managed in that unit. The weight for each impoundment in the IWEM surface impoundment site database was adjusted after the deletion of the six facilities with insufficient data, using a procedure analogous to the one described in Appendix A of the SIS report (U.S. EPA, 2001a). Details of the analysis and screening of surface impoundment facilities and units are fully described in the EPA SIS report (U.S. EPA, 2001a). Figure 4-7 shows the geographic locations of the 144 surface impoundment facilities in the SIS. At the scale of this map, the individual unit(s) at each facility are not shown.

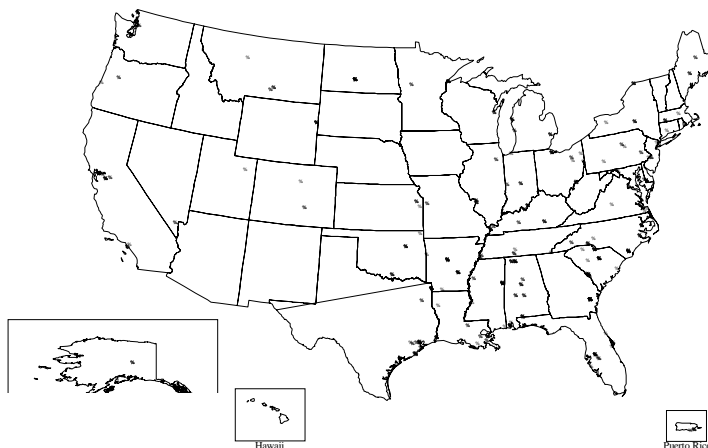


Figure 4-7. Geographic locations of surface impoundment WMUs.

Several additional steps were taken to compile the results of the SIS into the IWEM surface impoundment site database used for groundwater pathway modeling with the EPACMTP model. These steps are summarized here and are also described in the *IWEM Technical Background Document* (U.S. EPA, 2002b).

Surface impoundment sludge thickness represents the average thickness of accumulated sediment (sludge) deposits on the bottom of the impoundment, and this parameter is used to calculate the rate of infiltration from unlined and single-lined surface impoundments, as explained in the *EPACMTP Technical Background Document* (U.S. EPA, 2003b). For the purposes of EPACMTP modeling, it is assumed that the accumulated sediment consists of two equally thick layers, an upper unconsolidated layer and a lower consolidated layer (“filter cake”) that has been compacted as a result of the weight of the sediment above it, and therefore has a reduced porosity and permeability. Therefore, a total (unconsolidated + consolidated) sediment layer thickness of 0.2 m was assigned to each impoundment in the database (U.S. EPA, 2002b).

Unit-specific values for depth below ground surface were generally available from SIS data. If reported data on the depth of the base of the surface impoundment below ground surface

were missing or were reported as “unknown,” a depth below ground surface value of 0.0 m was assigned to each unit. An operational life of 50 years was assigned to each unit.

For surface impoundments, the EPACMTP model uses information on the distance to the nearest permanent surface water (that is, a river, pond, or lake) in the infiltration screening procedure presented in the *EPACMTP Technical Background Document* (U.S. EPA, 2003b). The data from the SIS indicated a distribution of values with a range of 30 to 5,000 m (3.1 mi) and a median value of 360 m (U.S. EPA, 2001a). The data reported in the SIS were used to assign a distance value to each surface impoundment unit in the IWEM surface impoundment site database. If a specific distance was reported, it was used; if the distance value was missing, reported as “unknown,” or “greater than 2,000 m,” a default value of 5,000 m was assigned to that impoundment.

Because the SIS includes only facility-specific data, dynamic linkages to two other data sources (HGDB and the HELP-modeled climatic database) are used to generate the additional, correlated input parameters required to perform the groundwater fate and transport modeling for each site. That is, for use in EPACMTP modeling analyses performed for the IWEM project, each of the 503 impoundments in the SIS database was classified according to the type of aquifer underlying the site and the closest climate center in order to provide links to the databases of hydrogeologic data (HGDB) and infiltration/recharge data (based on HELP water-balance modeling). This procedure was analogous to that done for the landfill site database (see Section 4.4.1.2). Additional details can be found in the *EPACMTP Technical Background Document* and the *EPACMTP Parameters and Data Document* (U.S. EPA, 2003a,b).

A professional geologist used the geographic location of the site in combination with USGS state-by-state aquifer maps (Heath, 1984) to assign each of the 503 surface impoundment units in the database to one of the 12 hydrogeologic environments in HGDB (Newell et al., 1989, 1990; U.S. EPA, 1997b) as follows:

- Determine the groundwater region of each site using Heath (1984); for any sites assigned to the “Alluvial Valleys” region, determine the larger region in which the alluvial valley is located.
- Assign the hydrogeologic setting using SIS data, aquifer and surficial geology information obtained using geographic information systems (GIS), and narratives and block diagrams from U.S. EPA (1987).
- Using the look-up table in Newell et al. (1990), determine HGDB hydrogeologic environment from hydrogeologic setting.

Because of the small scale and differences in registration of the GIS maps, in some cases the data sources would disagree. In these cases, the following hierarchy was used to determine environment: (1) SIS data; and (2) the most prevalent setting for GIS data coverages. The most prevalent setting was selected because the GIS coverages (Clawges and Price, 1999a,b,c,d; Miller and White, 1998) are of similar scale and accuracy, and therefore a predominance of evidence approach was the best way to address uncertainty in a particular data set. Because Newell et al. (1990) define two alluvial environments (6, River alluvium with overbank deposits,

and 7, River alluvium without overbank deposits), it was necessary to determine which environment an alluvial site falls into. The SIS soil layer information was used to distinguish between these two settings by determining whether there were significant fine-grained overbank deposits in the soil column. Quality assurance/quality control (QA/QC) measures included independent review of the assignments by other geologists with expertise in assigning settings. Any differences in assignment were then discussed and resolved by consensus.

The geographic location of the site was also used to assign each of the 503 impoundment sites in the database to one of the 102 climate centers for which HELP-modeled recharge rates are available (U.S. EPA, 1997b, 2003a). Unlike for landfills, for which HELP-modeled infiltration rates were used, the infiltration rates for surface impoundments are calculated internally by EPACMTP, as described in Section 4.4.2.4.

Additionally, because the groundwater temperature at each surface impoundment site is required to perform the groundwater fate and transport modeling but was not included in the SIS, these data were added during development of the IWEM surface impoundment site database in the same manner as was done for the landfill site database (see Section 4.4.1.2).

4.4.2.3 Summary of the Surface Impoundment Database Creation for the Current Modeling Analysis. Only one simple change was required to convert the IWEM surface impoundment site database into a database of surface impoundment sites appropriate for the modeling of surface impoundments used by the dyes and pigments industry: nine impoundments, located in Hawaii, Alaska, and Puerto Rico were deleted because the scope of the current study only includes impoundments located in the contiguous 48 states. Thus, there are 494 (not 503) impoundments included in the surface impoundment site database used for the groundwater pathway modeling conducted in support of the dyes and pigments listing determination.

4.4.2.4 Infiltration and Recharge Data Used for Each Surface Impoundment Liner Scenario. The modeling conducted for the surface impoundment scenario included analyses for each of three liner scenarios: no liner, clay liner, and composite liner. For all three liner scenarios, the default EPACMTP database of ambient regional recharge rates (generated by the HELP model) was used (U.S. EPA, 1994a, 1994b, 2002b, 2003a). The infiltration rates for unlined and single-lined surface impoundments are calculated internally by EPACMTP, as described in the *EPACMTP Technical Background Document* (U.S. EPA, 2003b). For composite-lined surface impoundments, the Bonaparte equation (Bonaparte et al., 1989) was used to calculate the infiltration rate, assuming circular (pin-hole) leaks with a uniform leak size of 6 mm² and using the distribution of leak densities (number of leaks per hectare) assembled from the survey of composite-lined units (TetraTech, 2001). The algorithms used to determine surface impoundment composite-liner infiltration rates are fully described in the *IWEM Technical Background Document* (U.S. EPA, 2002b).

4.4.2.5 Surface Impoundment Conceptual Model. In a surface impoundment, liquids and solids typically separate in a surface impoundment by gravity settling. Liquids from surface impoundments are removed by draining, evaporation, or flow through an outlet structure. Accumulated solids are typically removed by dredging during impoundment operation or at the time of closure. All surface impoundments used to manage dye and pigment industry wastes are

assumed to be onsite and used to manage only wastewaters. The impoundments were modeled as either a quiescent or aerated system. The sludges that accumulate in the surface impoundment are assumed to be managed in landfills. It is assumed that there are no direct liquid discharges to the surface due to overflows or structural failures. Therefore, the output from the surface impoundment source model provides air emissions as input for calculations of air concentrations and leachate concentration as input for groundwater pathways. After the operating period, the surface impoundment site is assumed to be closed and all constituents removed.

4.4.2.6 Surface Impoundment Characteristics. Before executing the surface impoundment source model, several calculations were performed to develop the values for parameters used to characterize surface impoundments (i.e., depth, volume, flow rate, residence time). The SIS (U.S. EPA, 2001a) was used to obtain these parameter values because it is the most recent survey of surface impoundments currently operating in the United States and the quality of the data in this survey has been thoroughly reviewed. When data for these parameters were available from the SIS, those data were used for this analysis; when data were not available from the SIS, values were estimated based on other SIS data.

When the SIS data set did not contain residence time for a particular unit, residence time was determined by randomly sampling from the distribution of residence times reported in the SIS. Before sampling from existing data points, however, the distribution was examined and the extreme values in the distribution were found to be unreasonable for sampling for this purpose. When the SIS data set contained neither flow rate nor residence time for a particular unit, flow rate was determined by using the randomly sampled residence times (determined as discussed above) and volume.

4.4.3 Tanks

To model risks associated with management of dye and pigment wastes in aerated and nonaerated treatment tanks, detailed information about tank characteristics was compiled. This section discusses the methods and data used to characterize aerated and nonaerated treatment tanks.

4.4.3.1 Selecting Representative Tank Units. The Industrial Subtitle D Survey (Schroeder et al., 1987) did not include tanks. Therefore, a tank database was developed for this analysis that compiled flow rates and tank volumes. The primary source for these data was EPA's 1986 National Survey of Hazardous Waste Treatment, Storage, Disposal, and Recycling Facilities (TSDR) Database (U.S. EPA, 1987). This database is the result of a comprehensive survey of 2,626 hazardous waste TSDR facilities that provided information concerning 1986 waste management practices and quantities. Responses were received from 2,322 facilities. The TSDR survey included a specific supplemental questionnaire concerning tanks used at each facility. Responses to this questionnaire provided tank information for about 18,773 tanks at 1,700 facilities.

A subset of the TSDR survey responses was available for facilities that received any quantity of waste from offsite sources. This subset of data contained information on 8,510 tanks located at 710 facilities (approximately 45 percent of all of the tanks contained in the TSDR survey). This reduced data set was used to characterize tanks for this analysis. The subset data

include a broad range of tank volumes ranging from less than 55 gal to more than 5 million gal. Though the available tank data were limited to onsite tanks, the data represent a wide range of tank sizes and types. This large data set is an excellent source of data to represent tanks in general, irrespective of whether they are located onsite or offsite, and whether or not the facility receives offsite wastes.

Table 4-5 shows the total number of tanks in the database, those culled from the database, and those remaining in the database that were used in this study. The totals add up to fewer than 8,510 because the database included 1,270 units that, based on process codes, were not actually tanks. The remaining 7,240 were categorized as treatment or storage tanks based on process codes, as described in Section 4.3.2.3; there were 472 tanks that specified both treatment and storage codes and that were therefore included in both treatment and storage data sets.

Table 4-5. Summary of Tanks Removed from TSDR Survey Database

Description	Treatment Tanks
Number of sites reporting WMU type in TSDR database	2,346
Culled sites: zero size or flow	464
Culled sites: tank covered or cover not specified	979
Culled sites: size \leq 55 gallons	6
Culled sites: outside contiguous 48 states	4
Total number of sites included in dyes and pigments listing risk assessment data set	893

Several criteria were used in guiding the development of the tanks database. These criteria were applied to the TSDR survey data to determine which tanks should be excluded from the data set:

- **Flow rate.** Only those tanks reporting nonzero flow rates were included in the analysis.
- **Open versus covered tanks.** Only treatment tanks were considered in the analysis; closed or covered tanks were dropped because this study is concerned only with Industrial D scenarios, and RCRA does not require covers for nonhazardous tanks.
- **Tank volume.** All tanks with a volume of 55 gal or less were excluded from the analysis. Because of their size, these smaller-volume containers should be classified as drums and not tanks. Additionally, two very large tanks (approximately 30 million gal), one aerated treatment and one nonaerated treatment, were reviewed because these tanks were many times larger than the next largest tanks and appeared to be nonrepresentative. The facility that owns

both tanks was contacted and reported that both tanks actually have volumes of 3 million gal, a value within the range represented by the other tanks in the database. Both values were corrected to 3 million gal for this analysis.

- **Location.** Tanks located outside the contiguous 48 states were excluded from the database.

4.4.3.2 Tank Characteristics. Industrial treatment tanks can be either quiescent or aerated/agitated. Examples of quiescent treatment tanks are clarifiers and filters (such as sand or mixed-media filters). In the absence of aeration, quiescent treatment tanks are still subject to small amounts of agitation during filling and emptying operations if they have above-surface intakes. Aeration or agitation in a wastewater treatment system transfers air to the liquid to improve mixing or increase biodegradation. The turbulence caused by aeration/agitation also enhances mass transfer to the air, thus increasing emissions. Therefore, for a given treatment volume, a facility with aerated tanks will have higher emissions than a facility with quiescent tanks.

To reflect emission characteristics associated with differences within the treatment tank category related to aeration intensity, three different tank categories were identified and modeled:

1. High-aeration treatment tanks
2. Low-aeration treatment tanks
3. Nonaerated treatment tanks.

The tanks in the database were sorted into these three categories using the WMU code reported for each unit. In the TSDR survey, the respondents were asked to provide a WMU code to describe the type of process for which each tank was used. The TSDR survey used a broad range of WMU treatment codes (including codes for incinerators and belt filter presses). Classification of treatment tanks was based on those processes listed in Table 4-6.

The treatment tank WMU codes were further evaluated to determine the level of aeration used. High (HI) aeration was assigned to tanks reporting processes that actively mix the liquid surface for the purpose of aeration or that add diffused air. Low (LO) aeration was assigned to tanks reporting processes that are likely to require mixing devices because of the addition of chemicals or other purposes. No (NO) aeration was used for tanks that are purposefully operated to minimize mixing or agitation (e.g., a clarifier). The aeration level assignments for each WMU code are shown in Table 4-6. The high- versus low-aeration classification is based on the nature of the process description associated with the various process codes:

- Equalization, cyanide oxidation, general oxidation, chemical precipitation, and chromium reduction all involve adding and mixing a chemical into the wastewater and then following the mixing process with a quiescent period. Tanks that use these processes were classified as LO aeration because the chemical addition and mixing involve more agitation than is used in a quiescent tank but involve no processes with intense agitation or forced air.

Table 4-6. TSDR Survey Wastewater Treatment Codes Used in Identifying Treatment Tanks

Process Code/Process	Aeration Level	Process Code/Process	Aeration Level
Equalization		Filtration	
1WT Equalization	LO	34WT Diatomaceous earth	NO
Cyanide oxidation		35WT Sand	NO
2WT Alkaline chlorination	LO	36WT Multimedia	NO
3WT Ozone	LO	37WT Other filtration	NO
4WT Electrochemical	LO	Sludge dewatering	
5WT Other cyanide oxidation	LO	38WT Gravity thickening	NO
General oxidation (including disinfection)		Air flotation	
6WT Chlorination	LO	43WT Dissolved air flotation	HI
7WT Ozonation	LO	44WT Partial aeration	HI
8WT UV radiation	LO	45WT Air dispersion	HI
9WT Other general oxidation	LO	46WT Other air flotation	HI
Chemical precipitation		Oil skimming	
10WT Lime	LO	47WT Gravity separation	NO
11WT Sodium hydroxide	LO	48WT Coalescing plate separation	NO
12WT Soda ash	LO	49WT Other oil skimming	NO
13WT Sulfide	LO	Other liquid phase separation	
14WT Other chemical precipitation	LO	50WT Decanting	NO
Chromium reduction		51WT Other liquid phase separation	NO
15WT Sodium bisulfite	LO	Biological treatment	
16WT Sulfur dioxide	LO	52WT Activated sludge	HI
17WT Ferrous sulfate	LO	54WT Fixed film--rotating contactor	LO
18WT Other chromium reduction	LO	57WT Anaerobic	NO
19WT Complete metals treatment	LO	58WT Other biological treatment	HI
Emulsion breaking		Other wastewater treatment	
20WT Thermal	NO	60WT Neutralization	LO
21WT Chemical	LO	61WT Nitrification	LO
22WT Other emulsion breaking	LO	62WT Denitrification	LO
Evaporation		63WT Flocculation and/or coagulation	NO
31WT Solar	NO	64WT Settling (clarification)	NO
Fuel blending		66WT Other wastewater treatment	LO
1FB Fuel blending	LO	Other processes	
		1TR Other treatment	LO

- Emulsion breaking includes two different processes. Thermal heating simply involves heating and letting the wastewater stand, whereas chemical emulsion breaking involves chemical addition and mixing followed by a quiescent period. Therefore, thermal emulsion breaking was classified as NO aeration and chemical emulsion breaking was classified as LO aeration. The category “other emulsion breaking” was classified as LO because the other processes in the emulsion breaking category ranged from NO to LO, so this represented a conservative default classification in the absence of more specific process data.
- Filtration processes are quiet and generally covered; therefore, these were classified as NO aeration. Many of these, in fact, were eliminated from the database because covered tanks, as a class, were removed.
- Air flotation processes all involve high-energy forced air operations and are, therefore, all classified as HI aeration.
- Oil skimming involves liquid-phase separation, which requires quiescent conditions; therefore, NO aeration was assumed for these processes. Similarly, liquid-phase separation processes were classified as NO aeration.
- Biological treatment processes are quite diverse and include HI aeration activated sludge processes and LO aeration film processes. The “other biological treatment” processes were classified as HI because the other processes in the biological treatment category ranged from NO to HI, and HI represents a conservative default classification in the absence of more specific process data.
- Finally, the “other wastewater treatment” process in the “other wastewater treatment” category and the “other processes” category were classified as LO aeration as a default because no process information can be inferred from the description.

The numbers of tanks included in each classification is summarized in Table 4-7.

Table 4-7. Numbers of Tanks by Classification

Tank Classification	Number
Aerated treatment tanks	620
HI aeration	29
LO aeration	591
Nonaerated treatment tanks	273
Total	893

Additional Tank Data Used for Imputation. In previous studies, additional data sources were identified to address tank-specific data gaps in the tanks database. These data included information collected in 1985 and 1986 during EPA site visits to aerated treatment systems in support of the development of RCRA air emission standards. In these studies, information on wastewater treatment systems at 54 facilities was collected. Data on the individual tanks (both aerated and nonaerated) were provided by the facilities during the site visits, including data on tank dimensions. The data on aerated tanks are summarized in RTI (1988) and Eichinger (1985). The data on nonaerated tanks were collected during the same site visits and are unpublished. Added to these data were five tanks from the TSDf background information document (BID; U.S. EPA, 1991). This resulted in a supplemental database of 49 tanks (13 with HI aeration, 9 with LO aeration, and 27 with NO aeration), presented in Table 4-8.

Table 4-8. Summary of Tank Size Information Collected in EPA Site Visits for RCRA Air Emission Standards

Type of Unit	Aeration	Type of Aerator	Volume (m ³)	Area (m ²)	Depth (m)
Aerated treatment tank	HI	Mechanical	108	27	4.0
Aeration tank	HI	Mechanical	112	34	3.4
Bubbling pit	HI	Diffused	453	74	6.1
Aerated treatment tank	HI	Mechanical	1,600	430	3.7
Aeration tank	HI	Diffused	1,666	159	10.5
Aeration tank	HI	Mechanical	3,367	910	3.7
Aeration tank	HI	Diffused	3,785	730	5.2
Aeration tank	HI	Diffused	4,542	618	7.4
Aeration tank	HI	Mechanical	5,678	931	6.1
Aeration tank	HI	Diffused	5,764	1,051	5.5
Aux. Aeration. tank	HI	Mechanical	21,804	4,459	4.9
Aeration tank	HI	Mechanical	26,546	5,806	4.6
Aeration tank	HI	Mechanical	41,261	11,241	3.7
Treatment tank	LO		30	13	2.4
Mixing tank	LO	Mechanical	68	9.3	7.3
Treatment tank	LO		76	26	2.7
Mixing tank	LO	Mechanical	112	34	3.4
No eq. basin	LO	Mechanical	191	84	2.3
So eq. basin	LO	Mechanical	240	109	2.2
Eq. basin	LO	Mechanical	681	200	3.4

(continued)

Table 4-8. (continued)

Type of Unit	Aeration	Type of Aerator	Volume (m ³)	Area (m ²)	Depth (m)
Treatment tank	LO		800	65	12.0
Eq. basin	LO	Mechanical	41,261	11,241	3.7
Gravitator	NO		5.6	1.8	3.1
Prefilter	NO		132	39	3.4
Final filter	NO		154	42	3.7
Backwash clarifier	NO		207	71	2.9
Clarifier	NO		283	46	6.1
Biosludge holding tanks	NO		300	66	4.6
Biosludge holding tanks	NO		300	66	4.6
Primary clarifier	NO		641	263	2.4
Digester	NO		819	117	7.0
Api separator	NO		836	457	1.8
Primary clarifier	NO		1,803	591	3.1
Biosludge thickener	NO		1,803	591	3.1
Clarifier/thickener	NO		2,504	410	6.1
Final clarifier	NO		2,513	687	3.7
Final clarifier	NO		2,513	687	3.7
Final clarifier	NO		2,513	687	3.7
Clarifier	NO		2,670	730	3.7
Clarifier	NO		2,670	730	3.7
Clarifier	NO		2,670	730	3.7
Clarifier	NO		3,065	804	3.8
Clarifier	NO		3,065	804	3.8
Clarifier	NO		3,065	804	3.8
Clarifier	NO		3,065	804	3.8
Ship's ballast water	NO		3,394	1,271	2.7
Final clarifier	NO		3,918	1,430	2.7
Ship's ballast water	NO		10,244	1,051	9.8
Solid waste disposal basin	NO		386,464	60,385	6.4

In addition to collecting these data, EPA contacted several tank vendors to establish a reasonable high end for tank capacity and depth based on design principles. As a result, a reasonable maximum capacity for an open, partially, or completely aboveground tank was defined to be approximately 3 million gal, and the depth of such a tank would not be expected to exceed 10 m (about 32 ft) (Kendall Smith, personal communication, AO Smith Industrial, March 16, 1999). These site visit tanks and hypothetical tanks were used only as a basis for imputing values and were not modeled in the analysis to maintain the integrity of the source database.

Estimation of Missing Data. The TSDR survey provided flow rate and tank volume data for use in characterizing tanks for this analysis. However, other key parameters, including depth, surface area, and height, also needed to be defined. In the absence of reported TSDR survey data, these parameters were calculated as described below. Other operating parameters (aeration parameters), which affect emission estimates but not dispersion, are discussed in Section 5.1.

The depth of the waste was imputed from the reported tank volume (or capacity). This was accomplished by developing a regression of log (depth) versus log (capacity) using data in the supplemental data set discussed below (49 tanks from the site visit/TSDR BID database plus TSDR BID tanks). Because the site visit data did not include any very small tanks or many very large tanks, a cube-shaped 55-gallon tank and a 3-million-gallon/32 ft deep tank (based on the vendor information) were included in the regression derivation. Regression lines were derived for aerated tanks (Equation 4-1) and nonaerated tanks (Equation 4-2), on the assumption that these might have different volume-to-depth relationships because aerated tanks may be shallower to facilitate aeration.

$$D = 10^{[0.1358 \times \log(V) + 0.2236]} \quad (4-1)$$

$$D = 10^{[0.1334 \times \log(V) + 0.1657]} \quad (4-2)$$

where

$$\begin{aligned} D &= \text{depth (m)} \\ V &= \text{volume (m}^3\text{)}. \end{aligned}$$

However, as can be seen in Figure 4-8, the two regressions were nearly identical. Therefore, a single regression was developed using all 49 tanks from the site visit/TSDR BID database plus the hypothetical tanks, as follows:

$$D = 10^{[0.1057 \times \log(V) + 0.2804]} \quad (4-3)$$

Comparisons of this regression with regressions done without one or both hypothetical tanks indicate that the hypothetical tanks do not unduly dominate the regression.

This equation was then examined for the reasonableness of the depths predicted. Using Equation 4-3, 60-gallon tanks (the smallest tanks in the database) are approximately 1.9 m

(6.2 ft) deep and about 39 cm (15.4 in) in diameter. This seemed unrealistically tall and narrow; consequently, for very small tanks, a second equation was derived from the assumption of a cube-shaped tank:

$$D = V^{0.333} \quad (4-4)$$

For tanks of approximately 10 m³, Equation 4-3 predicts approximately cube-shaped tanks; therefore, Equation 4-4 is used for tanks smaller than 10 m³.

The largest tank in the TSDR database in the NO and LO aeration categories is 25,000 m³, and the projected depth for this tank using Equation 4-3 is 5.6 m (18 ft), which is acceptable for mixing tanks. The largest tank in the TSDR database in the HI aeration category is 23,000 m³, and the projected depth for this tank is 5.5 m (18 ft). In evaluating the predicted depth of HI aeration tanks, the eight mechanically aerated tanks from the site visits were considered. The maximum depth from these data was 6.1 m (20 ft), and even this appeared to be an outlier compared to the other HI aeration, mechanically aerated tanks. Data for the other seven site visit tanks showed depths ranging from 3.35 m (11 ft) to 4.88 m (16 ft). The mid-range of the latter depths is approximately equivalent to a 1,000 m³ tank, as evaluated using Equation 4-3. Therefore, for HI aeration tanks greater than 1,000 m³, a random depth was assigned using a uniform distribution with endpoints of 3.5 m and 4.8 m.

Table 4-9 summarizes the methods used to make an initial estimate of tank depth for each type of tank. Although these methods were intended to represent the actual relationship between volume and depth as closely as possible, they imply a certain precision that is unrealistic. In fact, there will be variation in the dimensions of tanks of the same volume. To address that variation, a random variation was applied to these initial estimates using a normal distribution with a mean of 1 and 90 percent of the values between 0.8 and 1.2. The initial depth estimate was multiplied by this random factor to obtain a final depth estimate used in this analysis.

Surface area data were not provided in the TSDR survey. In the absence of these data, surface area for each of the TSDR tanks was calculated by dividing tank volume by depth.

The height of the top of the tank above the ground is needed for dispersion modeling. Height is related to depth, but not necessarily equal to depth, as tanks may be partially in the ground. In the absence of height data being reported in the TSDR survey, height was imputed from depth using a two-step process:

1. A number was selected at random from 0 to 20 (uniform distribution).
2. If this number was less than the depth in meters + 0.5 meters, it was used as the height. If it was greater than the depth in meters + 0.5 meters, height was set to depth + 0.5 m.

None of the tank depths imputed were greater than 9.5 m; therefore, none of the heights aboveground were more than 10 m (9.5 + 0.5) using this method; 10 m above the ground is the realistic maximum height from a structural point of view, according to tank vendor contacts.

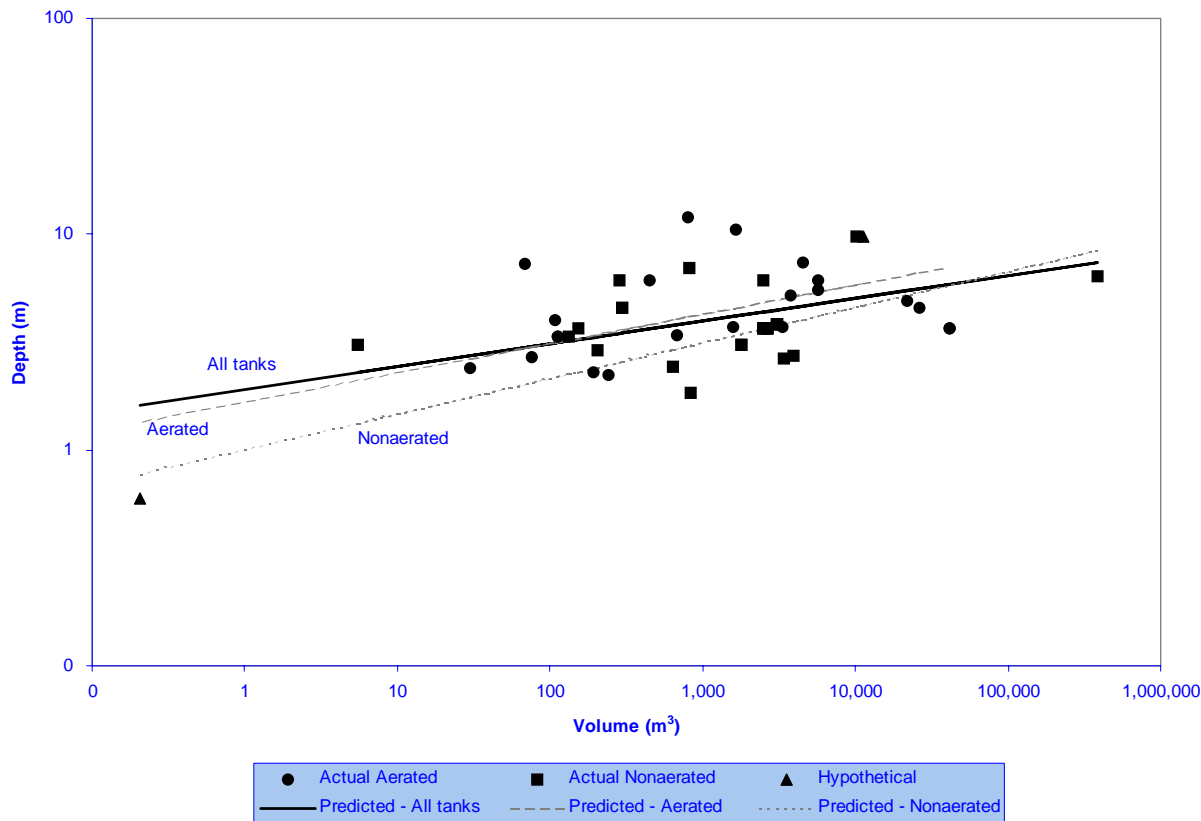


Figure 4-8. Comparison of tank depth regression lines.

Table 4-9. Summary of Depth Imputation Techniques

Tank Type	Volume Range (m³)	Imputation Technique
HI aeration	<10	Equation 4-3
	10-1,000	Equation 4-4
	>1,000	Uniform distribution from 3.5 to 4.8 m
LO aeration	<10	Equation 4-3
	≥10	Equation 4-4
NO aeration	<10	Equation 4-3
	≥10	Equation 4-4

This approach establishes percentages of tanks of certain depths that will be all aboveground vs. partially or completely in ground:

- For 10 m tanks, about one-half are partly or completely in the ground (when the random selection is between 0 and 10; as the random number increases from 0 to 10, more and more of the tank depth is aboveground, until, at 10, all of it is), and about one-half are completely aboveground (when the random pick is between 10 and 20).
- For 1 m tanks, about 5 percent are partly or completely in the ground (random numbers from 0 to 1) and 95 percent are completely aboveground (random numbers from 1 to 20).
- For 5 m tanks, 25 percent are partly or completely in the ground (random numbers from 0 to 5) and 75 percent completely aboveground (random numbers from 5 to 20).

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5.0 Exposure Point Concentrations

Exposure point concentrations are constituent concentrations at the location in the environment where a receptor may be exposed. To determine constituent concentrations in environmental media with which a receptor comes in contact (e.g., groundwater, air), several computer-based models and sets of equations are used. Generally, these include source models and fate and transport models.

For this risk assessment, it was assumed that dye and pigment wastes are deposited in landfills, surface impoundments, and tanks (WMUs are described in detail in Section 4.0). Chemical constituents found in dye and pigment waste are released from these WMUs into the surrounding environment. Releases to the atmosphere occur through volatilization of vapors from all three WMU types. Leachate is formed and migrates to groundwater from both landfills and surface impoundments. These materials are then transported in air and groundwater in the environment immediately surrounding the WMU.

The following subsections describe the models and equations used to calculate exposure point concentrations and their application in this risk assessment. Section 5.1 describes the source partition models used to predict environmental releases of constituents from WMUs. Section 5.2 discusses the fate and transport models used to predict ambient air, groundwater, and indoor air concentrations due to showering.

Greater detail is provided in appendices to this document:

- Appendix D, Source Data, presents the source partition model input parameter values used in this risk assessment for landfills, surface impoundments, and tanks.
- Appendix E, Groundwater Modeling Parameters, presents the input values or distributions used in the groundwater modeling.
- Appendix F, Unitized Air Concentrations, provides details on the unitized air concentrations (UACs) for this risk assessment.
- Appendix G, Exposure Equations, documents the algorithms used to calculate exposure point concentrations for air and water.

5.1 Source Modeling of Constituent Releases

Source partition models were used to estimate environmental releases of constituents from landfills, surface impoundments, and tanks. Each WMU has different release mechanisms that determine the environmental media affected:

- Wastes managed in **landfills** can release COCs as vapors to the air or as leachate to the groundwater. In this assessment, it was assumed that daily cover prevents release of particulates from the landfill surface.
- Release mechanisms from **surface impoundments** include volatilization to the air and leaching to groundwater. Because surface impoundments contain liquid waste, particulate emissions are not released and do not occur from this WMU.
- Wastes managed in **tanks** can release COCs into the atmosphere via volatilization. Because tanks contain liquid waste, particulate emissions are not released and do not occur from this WMU. Tanks were assumed not to leak or to result in direct releases to the groundwater or soil.

The source models use information for a specific WMU (e.g., surface area), constituent, environmental setting (e.g., precipitation, temperature), and waste stream to estimate environmental release of COCs for each release mechanism. Because the purpose of this assessment was to calculate mass loadings that are protective of human health and the environment, specific waste concentrations were not initially used in the source models. Instead, the models were executed using a unit mass load (e.g., 1 kg/yr). The results of these source model runs were then used to calculate target mass loads in the WMUs as described in Section 9.

5.1.1 Landfill Model

The landfill model was developed to approximate the dynamic effects of the gradual filling of active landfills. The modeled landfill is divided into equal-volume vertical cells running from the site surface to the bottom of the landfill, each sized so that it requires 1 year to fill. Waste mass is added gradually, forming layers of waste. After 1 year, the cell is full and covered with a clean final soil cover. Then, the next cell begins to fill, and so on until the landfill reaches maximum capacity. Results for the landfill as a whole are then obtained by aggregating results for each single cell to account for the time that each cell in the landfill was filled. For example, the results for the landfill at the end of year 3 are a summation of results for the first cell filled at year 3 in the single-cell simulation, the second cell filled at year 2, and the third cell filled at year 1.

The active life of a landfill is assumed to be 30 years; therefore, 30 cells were modeled by the landfill model. The model estimates environmental releases from the landfill beginning in the first year of operation. The landfill fills for 30 years, after which time no more constituent mass is being added and the model accounts for continuous loss of constituent mass. This modeling process continues until 1 percent of the peak constituent mass remains or until 200 years, whichever comes first.

Other assumptions made in modeling landfills include

- The landfill is assumed to be below grade or to have a proper berm so that all precipitation that falls onto the landfill is evapotranspired, percolates as infiltration, or increases the moisture content in the unit. That is, stormwater runoff, runoff, and erosion are assumed not to occur.
- Each landfill cell can be approximated as a soil column consisting of two homogeneous zones, as shown in Figure 5-1: soil cover and landfill waste. Each zone can be approximated as having homogeneous porous media whose properties are uniform in space and time within the zone but may differ between zones.
- All fluxes (air emissions and leachate) are expressed as averages (g/m^2) over the total surface area of the landfill despite the fact that the footprint of the filled cells is smaller than the total area of the landfill until the last year of operation.
- Waste is added to the landfill cell in layers. A waste layer, for the purposes of the model, is simply a zone wherein initial concentrations are assumed to be uniform. Waste layers are conceptualized as being formed over time by the dumping of loads of waste (e.g., via dump truck) next to one another in the landfill cell until a waste layer of uniform depth is eventually formed. At this point, a new layer is started.
- The first-order chemical and biological loss processes that occur in the entire landfill, including cover soil and waste, include anaerobic biodegradation (i.e., decomposition processes that occur in an oxygen-free environment) and hydrolysis.
- The annual average infiltration rate is assumed to be unaffected by the cover soil (i.e., cover soil is permeable).

As shown in Figure 5-1, waste is applied in layers within a single cell in the landfill. Because a cell takes 1 year to fill, the time required to lay down one layer is simply 365 days divided by the number of layers in a cell, where the number of layers in a cell is a user-specified number. The depth of a waste layer is determined by dividing the depth of the waste zone by the number of layers in the cell. The surface area of a cell is calculated internal to the model and is equal to the load (Mg/yr) divided by the depth of the waste layer (m) times the bulk density of the waste (g/m^3) and a units conversion factor. The number of years the landfill operates then is the total surface area of the landfill divided by the surface area of a single waste cell. For this analysis, these parameters were derived to result in a 30-year landfill lifetime.

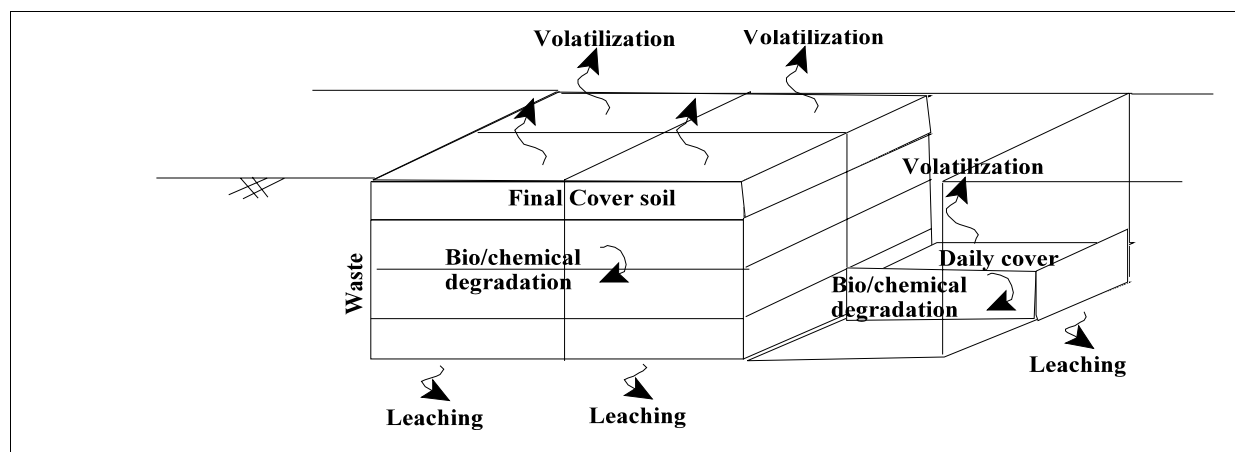


Figure 5-1. Illustration of landfill with six cells and three waste layers.

5.1.1.1 Model Overview. The landfill model consists of several interacting submodels or algorithms, the most significant of which are the Generic Soil Column Model (GSCM) and the hydrology model. These two primary submodels are discussed below, followed by a discussion of landfill liner scenarios.

Generic Soil Column Model. The GSCM is the fundamental “building block” of the landfill partition model; it describes the dynamic, vertical fate and transport of constituent within a single landfill cell. As stated previously, the GSCM simulation of the processes in the landfill cell begins in the first year of operation and continues through the filling of the cell (1 year) and thereafter until less than 1 percent of the cell’s peak constituent mass remains or 200 years, whichever occurs first. The complete time series of the GSCM emissions simulations for the first cell (perhaps 200 years) is stored by the landfill model and is then used to represent new cells being filled in subsequent years. The assumptions and limitations of the GSCM are summarized here and are described in full in U.S. EPA (1999):

- The medium modeled, whether soil, waste, or a soil/waste mixture, can be approximated for modeling purposes as an unconsolidated, homogeneous, porous medium.
- Internal loss processes, which include anaerobic biodegradation and hydrolysis, can be considered to proceed in accordance with first-order reaction kinetics.
- The contaminant partitions to three phases: adsorbed (solid), dissolved (liquid), and gaseous. This partitioning of constituent mass is similar to the methods described in Jury et al. (1983, 1990).
- Contaminant partitioning between sorbed and aqueous phases is reversible and linear. Furthermore, the partition coefficient is unaffected by changes in

concentrations or environmental conditions (e.g., pH, temperature) during the model execution.

- Contaminant partitioning between aqueous and gaseous phases can be described by Henry's law. The gaseous-phase constituent volatilizes from the surface of the landfill to the air.
- The chemical is transported in one dimension through the soil column, a method similar to those methods described in Jury et al. (1983, 1990) and Shan and Stephens (1995).
- Formation of chemical species by chemical or biological processes in the landfill (e.g., daughter products) is not considered.
- Leaching of aqueous-phase constituent mass occurs by advection or diffusion from the bottom of the WMU or vadose zone.

Hydrology Model. The hydrology model provides estimates of daily soil moisture, runoff, and evapotranspiration, while infiltration through the landfill and soils surrounding the landfill (recharge) is estimated by the EPA's Hydrologic Evaluation of Landfill Performance (HELP) model. A composite approach to modeling hydrologic processes was adopted to simulate multiple liner scenarios required for this analysis that the current GSCM model does not support. The approach to modeling liner scenarios is described below. Runoff is assumed to be zero for the below-grade or properly bermed landfill. These daily estimates are then used by the GSCM algorithm in its daily time step to build up the annual average output variable values. Details for the hydrology algorithms incorporated into the landfill partition model are described in U.S. EPA (1999). Details on the use of HELP to generate infiltration and recharge rates for landfills are provided in the EPACMTP Parameters and Data Background Document (U.S. EPA, 2003b).

Liner Scenario. The landfill model does not simulate liners; therefore, some additional data and simplifying assumptions were used for the liner scenarios. The primary limitations of the landfill model for liner scenarios are as follows: (1) the hydrology model relies on a surface water balance and does not account for any constraints on infiltration rates at depth in the soil column, such as would occur with a liner; and (2) the GSCM functionality is not sufficiently robust to take into account that sorption sites in a liner are likely to be exhausted long before sorption sites within the waste material itself. Specifically, in reality, dissolved material in the leachate would pass through the liner virtually unchanged (to the extent that there is infiltration across the liner), and because exhaustion of available surface sites would result in near-zero net sorption, the only loss due to sorption would be retardation and decay during retardation (unlike in the waste zone, where active net removal due to sorption presumably does occur).

To compensate for the limitations of the hydrology model, EPA's HELP model (U.S. EPA, 1994a,b) was used to estimate the infiltration rates for the unlined scenario and for a lined scenario with a compacted clay liner. Infiltration rates also were compiled for composite liner scenarios that employ a geomembrane in conjunction with compacted or geosynthetic clays. These rates were compiled from monthly average leak detection system flow rates for two

different distributions of composite liner types for similarly configured landfill cells reported by TetraTech (2001). (The composite liner scenarios are detailed in Section 4.) These infiltration rates were then used as inputs to the landfill model, which contains an option to use a user-specified infiltration rate and bypass the model's internal hydrologic calculations. To address the liner kinetics and sorption limitations of the GSCM, it was simply assumed that no liner exists. That is, chemical concentration predicted in the leachate is the same concentration that would result from a landfill with no liner, but that has an infiltration rate equal to the HELP-estimated liner infiltration rate. Because, as discussed above, this assumption does not account for some decay during retardation that would occur in a liner, it is a conservative estimate.

5.1.1.2 Model Input Parameters. The input parameters for the landfill model can be categorized into physical, operating, meteorological, soil, vadose, bulk waste, waste stream, and chemical input parameters. Details pertaining to the parameter values used in the landfill partition model are presented in Appendix D.

5.1.1.3 Landfill Model Results. The landfill model was used to calculate emission rates for volatile emissions to the air, which were then used as inputs in the fate and transport models. Results were generated for 10,000 runs for each of the chemicals. Each run was identified by landfill WMU, meteorological site, and a run ID that corresponded to one of the 10,000 Monte Carlo iterations.

5.1.2 Tank Model

The tank model simulates time-varying releases of constituents to the atmosphere. The tank unit has only volatile emissions (no particulates) and is assumed to have an impervious bottom so that there is no contaminant leaching. Therefore, the tank model calculates only release of vapor to the aboveground pathways. The tank model is a quasi-steady-state model, and the emissions occur only while the unit is operating. Volatile emissions were calculated for 50 years, which is the specified years of operation for tanks in this analysis.

Tanks may be either aerated or quiescent and may have some degree of solids settling. For aerated units, suspended solids in the influent waste primarily pass through the system with little solids settling (depending on the degree of agitation). For quiescent units, solids settling and accumulation may be significant. When this occurs, the unit has to be cleaned or dredged to remove the accumulated solids.

5.1.2.1 Model Overview. The tank model used in this assessment consists of the liquid and sediment components of the Aerated Tank Module. The liquid compartment model

- Uses a mass balance approach, taking into consideration contaminant removal by volatilization, biodegradation, hydrolysis, leaching, and partitioning to solids
- Estimates volatilization rates for both aerated and quiescent surfaces
- Estimates suspended solids removal (settling) efficiency
- Estimates temperature effects.

The model is a quasi-steady-state model that relies on monthly time steps. Quasi-steady-state refers to the fact that the model employs time steps in its mathematical solution, but within these monthly time steps, steady-state assumptions are made. Each month, the model updates certain parameters based on average monthly environmental conditions (temperature, windspeed, precipitation, and evaporation). It is then assumed that the system equilibrates instantaneously to these new conditions, and a steady-state solution is obtained for that month. The resulting 12 monthly values for all outputs are then averaged and reported as annual averages.

Key Assumptions. The general model construct can be useful for a wide variety of WMU applications. For this analysis, the following assumptions were used:

- Two-compartment model: a “mostly” well-mixed liquid compartment and a well-mixed sediment compartment, which includes a temporary accumulating solids compartment
- First-order kinetics for volatilization in the liquid compartment
- First-order kinetics (e.g., rates of change) for hydrolysis in both liquid and sediment compartments
- First-order kinetics for biodegradation with respect to both contaminant concentration and biomass concentration in liquid compartment
- First-order kinetics for biodegradation in the sediment compartment
- Darcy's law for calculating the infiltration rate
- First-order kinetics for solids settling
- First-order biomass growth rate with respect to total biological oxygen demand (BOD) loading
- First-order biomass decay rate within the accumulating sediment compartment
- No contaminant in precipitation/rainfall
- Linear contaminant partitioning among adsorbed solids, dissolved phases, and vapor phases
- Daughter products are not included in the model; any constituents generated as a reaction intermediate or end product from either biodegradation or hydrolysis are not included in the model output.

Because of the simplicity of the biodegradation rate model employed and the use of Henry's law partition coefficients, the model is most applicable to dilute aqueous wastes. At higher contaminant concentrations, biodegradation of toxic constituents may be expected to exhibit zero-order or even inhibitory rate kinetics. For waste streams with high contaminant or

high total organic concentrations, vapor-phase contaminant partitioning may be better estimated using partial pressure (Raoult's law) rather than Henry's law.

Methodologies. The treatment tank is divided into two primary compartments: a "liquid" compartment and a "sediment" compartment. Mass balances are performed on these primary compartments at time intervals small enough that the hydraulic retention time in the liquid compartment is not significantly affected by the solids settling and accumulation. Figure 5-2 provides a general schematic of a model construct.

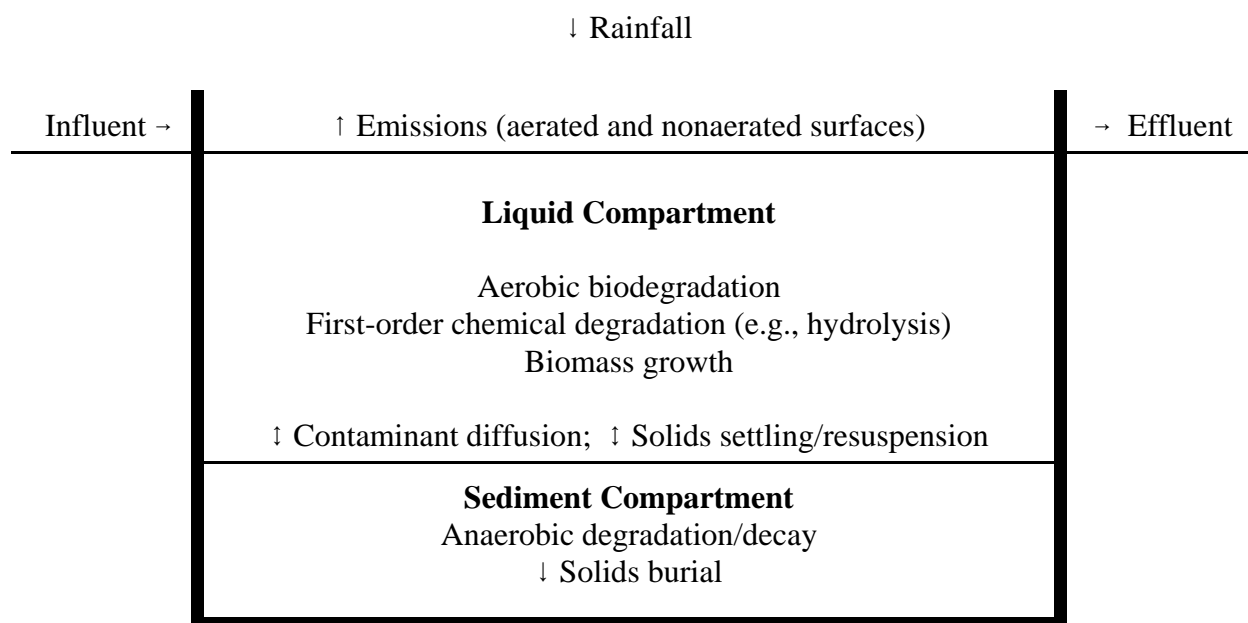


Figure 5-2. Schematic of general model construct for tanks.

In the liquid compartment, there is flow both into and out of the WMU. Solids generation occurs in the liquid compartment as a result of biological growth; solids destruction occurs in the sediment compartment as a result of sludge digestion. The suspended solids concentration within the WMU is assumed to be constant throughout the WMU. However, some stratification of sediment is expected across the length and depth of the WMU so that the effective total suspended solids (TSS) concentration within the tank is assumed to be a function of the WMU's TSS removal efficiency rather than equal to the effluent TSS concentration. The liquid (dissolved)-phase contaminant concentration within the tank, however, is assumed to be equal to the effluent dissolved-phase concentration (i.e., liquid is well mixed). Consequently, the term "mostly well mixed" is used to describe the liquid compartment.

The steady-state, mass balance assumptions on which the model is based are summarized as follows:

Constituent Mass Balance in Liquid Compartment. In the liquid compartment, there is flow both into and out of the WMU.¹ Within the liquid compartment, there is contaminant loss through volatilization, hydrolysis, and biodegradation. Additionally, contaminant is transported across the liquid/sediment compartment interface by solids settling and resuspension and by contaminant diffusion.

Constituent Mass Balance in Sediment Compartment. Within the sediment compartment, there is contaminant loss through hydrolysis and biodegradation. Additionally, contaminant is transported across the liquid/sediment compartment interface by solids settling and resuspension and by contaminant diffusion.

Solids Mass Balance in Liquid Compartment. Sedimentation and resuspension provide a means of sediment transfer between the liquid and sediment compartments. Sedimentation and resuspension are assumed to occur in the quiescent areas. For systems in which biodegradation occurs within the liquid compartment, there is also a production of biomass associated with the decomposition of organic constituents.

Solids Mass Balance in Sediment Compartment. In the sediment compartment, as in the liquid compartment, sedimentation and resuspension provide a means of sediment transfer between the liquid and sediment compartments. In the sediment compartment, however, there is some accumulation of sediment during the time step. This sediment accumulation is also referred to as sediment burial, and the rate of sediment accumulation is determined by the burial velocity. The primary output of the tank model is the annual average volatilization rate for each constituent.

5.1.2.2 Model Input Parameters. The input parameters for the tank model can be categorized into tank, meteorological, bulk waste, and chemical input parameters. Details pertaining to the parameter values used in the tank model are presented in Appendix D.

5.1.2.3 Tank Model Results. The tank model was used to calculate the total emission rate of volatile emissions to the air, which was then used in the multipathway fate and transport models. Results were generated for 10,000 runs for each of the chemicals. Each run was identified by tank WMU, meteorological site, and a run ID that corresponded to one of the 10,000 Monte Carlo iterations.

5.1.3 Surface Impoundment Model

The surface impoundment model simulates time-varying releases of constituent to the atmosphere. The surface impoundment unit is the same as the tank unit, but the bottom of the unit is assumed to be pervious so that contaminant leaching can occur. It is assumed that there are no direct liquid discharges to the surface due to overflows or structural failures. Therefore, the output from the surface impoundment model provides air emissions as input for calculations of air concentrations, and leachate concentration as input for groundwater pathways. The model

¹ There is also a leachate flow to the sediment compartment and out the bottom of the WMU for surface impoundments (see Section 5.1.3).

is a quasi-steady-state model, and the emissions occur only while the unit operates. The constituent releases are calculated for 50 years, which is the specified number of years of operation for surface impoundments in this analysis. After the operating period, the surface impoundment site is assumed to be closed and all constituents removed.

The surface impoundment model used in this risk assessment consists of the liquid and sediment components of the Surface Impoundment Module. Because of similarities in mass balance and transport between tanks and surface impoundments, many of the equations and computer code used in the Aerated Tank Module are identical to those used in the Surface Impoundment Module. In this analysis, the tank model and surface impoundment model are the same for the liquid compartment; consequently, that discussion is not repeated in this section. See Section 5.1.2 for a full description of the volatile emissions calculations.

5.1.3.1 Model Overview. In addition to volatile emissions, the surface impoundment model calculates the average annual infiltration rate and the associated average annual leachate contaminant flux rate. Groundwater infiltration is assumed to be driven by the hydrostatic pressure head produced by the wastewater in the unit; when the unit ceases operation, no additional contaminant is assumed to leach from the source because the unit is closed and all liquid and sediment removed. Annual liquid infiltration rates and contaminant leachate flux rates are both calculated at the base of the unit and are output for use in the groundwater modeling.

Surface Impoundment Vadose Zone Infiltration. The leachate flux rate is calculated by multiplying the dissolved contaminant concentration in the sediment porewater by the infiltration rate of leachate flow. The infiltration algorithm estimates infiltration rates as a function of liquid depth in the WMU, the thickness and hydraulic conductivity of the sediment compartment (considered as two layers, unconsolidated and consolidated), either a native soil layer immediately below the WMU that has been clogged by infiltrating sediment or a compacted soil (clay) liner, and an unclogged native soil layer below the clogged layer or soil layer. A schematic diagram of the conceptual model for the infiltration rate algorithm is shown in Figure 5-3. An iterative method is used to converge on the infiltration rate satisfying the physical requirement that the hydraulic pressure head at the groundwater table equal the atmospheric pressure.

Three different liner scenarios were investigated: no liner, a compacted soil (clay) liner, and a composite liner. The surface impoundment model's internal infiltration rate algorithm was used for the no liner and clay liner scenarios. The only difference in the no liner and clay liner scenarios is that the depth and hydraulic conductivity of the clogged native material soil layer is replaced with the depth and hydraulic conductivity of the compacted clay liner. The surface impoundment model does not simulate composite liners; therefore, infiltration rates for liner scenarios employing a composite liner were compiled from monthly average leak detection system flow rates using a GM-GCL liner (see Section 4 for details) for similarly configured surface impoundments reported by TetraTech (2001). These infiltration rates were then specified as input to the surface impoundment model, and the internal infiltration rate algorithm was bypassed. The surface impoundment model estimates the leachate concentration at the sediment/liner (or clogged native material) soil layer. This concentration of the leachate exiting the liner layer was assumed to be the same concentration calculated by the model (i.e., the same

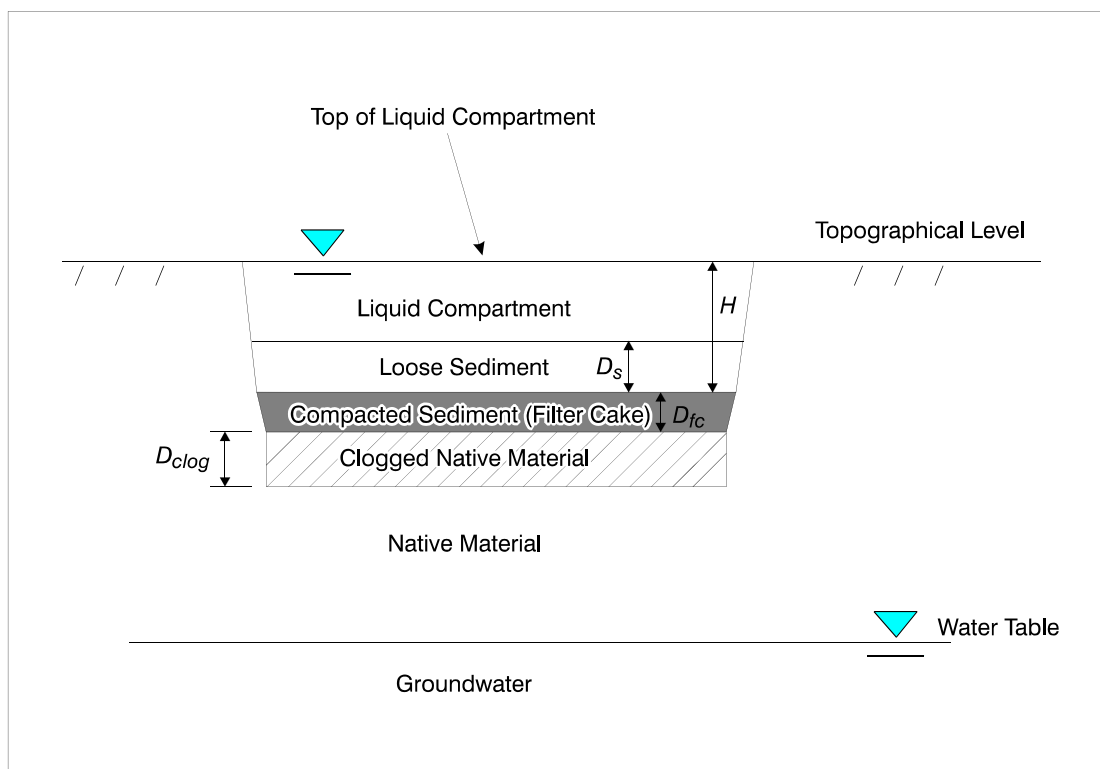


Figure 5-3. Conceptual model for surface impoundment infiltration algorithm.

as the concentration entering the liner layer). This assumption does not account for some decay during retardation that may occur in a liner, and therefore, it is a conservative assumption.

5.1.3.2 Model Input Parameters. The input parameters for the surface impoundment model can be categorized into physical and operating, meteorological, vadose, bulk waste, and chemical input parameters. Section 4.3 presents a detailed discussion of the location-dependent parameters. Details pertaining to the parameter values used in the surface impoundment partition model are presented in Appendix D.

5.1.3.3 Surface Impoundment Model Results. The surface impoundment model was used to calculate the total emission rate of particulate and volatile emissions to the air, which was then used as input to the multipathway fate and transport models. Results were generated for 10,000 runs for each of the chemicals. Each run was identified by surface impoundment WMU, meteorological site, and a run ID that corresponded to one of the 10,000 Monte Carlo iterations.

5.2 Fate and Transport Modeling

This section describes the methodology and the models that were used to predict the fate and transport of chemical constituents in the environment.

Once a constituent is released from a WMU, it can move through the air and groundwater. This transport into the environment may enable people and wildlife to be exposed to the released constituent. The purpose of the fate and transport modeling performed for this assessment was to estimate the concentration of a constituent in environmental media (i.e., air and water) at certain locations around a WMU where individuals or wildlife may be located. To predict a contaminant's movement through these media, fate and transport models are employed. Fate and transport models typically used by EPA are either a series of computer-based algorithms or sets of equations that predict chemical movement due to natural forces. These fate and transport models integrate information on a site's geology, hydrology, and meteorology with chemical, physical, and biological processes that can take place in the environment. The result is a simulation of chemical movement in the environment and a prediction of the concentration of a constituent at a certain point called the "exposure point." The following fate and transport models were used for this analysis:

- Air dispersion model
- Groundwater model
- Indoor air model (predicts the exposure to contaminants due to showering with contaminated groundwater).

These three models and the general framework for performing the fate and transport modeling are described in the following sections. Section 5.2.1 discusses the air dispersion modeling. Section 5.2.2 presents the model and methodology for predicting contaminant movement in groundwater, and Section 5.2.3 discusses the methodology for calculating indoor air concentrations due to domestic use of contaminated groundwater.

5.2.1 Air Pathway

Dispersion modeling involves using a computer-based set of calculations to estimate ambient ground-level constituent concentrations associated with constituent releases from a WMU. The dispersion model uses information on meteorology (e.g., windspeed, wind direction, temperature) to estimate the movement of constituents through the atmosphere. Movement downwind is largely determined by windspeed and wind direction. Dispersion around the centerline of the contaminant plume is estimated by empirically derived dispersion coefficients that account for movement of constituents in the horizontal and vertical directions.

The air concentrations for this analysis were based on the air dispersion factors, called UACs, from the Industrial Waste Air (IWAIR) model (U.S. EPA, 2002). These UACs are specific to a location, WMU surface area, and distance from the WMU. For this assessment, these parameters were set as follows:

- **Location.** EPACMTP Climate Center locations were used to place landfills and surface impoundment locations, as described in Section 4.0. For tank locations, the locations contained in the TSDf survey tanks database were used, as described in Section 4.0. These locations were then cross-referenced to the meteorological regions developed for IWAIR.

- **Surface area.** Surface areas were obtained from several sources: the Industrial D database for onsite and offsite industrial landfills, the distribution of sizes of municipal landfills developed for EPACML for municipal landfills, the Surface Impoundment Survey for surface impoundments, and the TSDF survey tanks database for tanks. Height data for tanks were also obtained from this database. Details about WMU surface area determination are presented in Section 4.0.
- **Distance.** The TSDF survey (U.S. EPA, 1991a) provides a basis for developing a distribution of distances ranging 50 to 500 m, with a median of 300 m; this distribution was developed for the paints listing risk assessment (U.S. EPA, 2001). For the dyes and pigments risk analysis, this distance distribution was converted into a discrete distribution containing only the exact distances for which UACs were available in IWAIR.

As directed by EPA, the dyes and pigments assessment is a nationwide analysis that assumes dyes and pigment manufacturing wastes could be disposed of anywhere in the United States. UAC values previously developed for the IWAIR model were used for this analysis because these values were developed for both ground-level and elevated sources; they are up to date regarding versions of the ISC model and meteorological data; and they were calculated for 60 regions of the United States, which were rigorously developed and reviewed as part of IWAIR development. In using the IWAIR UAC values, some uncertainty was introduced into the analysis because the UACs are regional and not site specific; they are maximum concentration values and thus introduce a high bias into the air pathway; and the interpolation routines did not exactly fit the characteristics of the sources modeled in this analysis. These uncertainties are relatively small.

To calculate UACs for this analysis, the IWAIR UAC for the corresponding location and distance were selected and the same interpolation schemes used in IWAIR were used to interpolate to the desired area (for landfills and surface impoundments) or area and height (for tanks). For landfills and surface impoundments with surface area values less than the values in the IWAIR UAC database, the smallest area available in IWAIR was used to determine UACs. For landfills with surface area values greater than the values in the IWAIR UAC database, the largest area available in IWAIR was used to determine UACs. IWAIR was not designed to evaluate tanks specifically; however, it does contain a set of UACs for use with elevated sources. These UACs can be used for any elevated sources, much as the ground-level UACs are used for any ground-level sources (e.g., landfills, surface impoundments) because the ISC dispersion model for area sources only considers surface area and height in estimating ground-level air concentrations. The limitation in using the existing IWAIR elevated source UAC values is that, because IWAIR does not include tanks, the range of surface areas covered by IWAIR's UACs does not include the lower end of the surface areas for the tanks sizes evaluated in the dyes and pigments analysis. To calculate UACs for the smaller tanks, UACs between zero and the minimum were interpolated for the available IWAIR areas. The technical rationale supporting this approach is contained in Appendix F.

The resulting UAC values were then imported to the exposure and risk model and used with volatile emission rates calculated by the source models to determine ambient air

concentrations. For each run ID, an air concentration of volatile constituents was calculated. These were averaged over the exposure duration using a time period centered around the peak value, and this average was used to estimate exposures.

5.2.2 Groundwater Pathway

The groundwater pathway was modeled for this analysis to determine the receptor concentrations resulting from the release of waste constituents from the WMU. The release of a constituent occurs when liquid that percolates through the WMU generates leachate, which can infiltrate from the bottom of the WMU into the subsurface. For landfills, the liquid percolating through the landfill is from precipitation and water in the waste. For surface impoundments, the liquid is primarily the wastewater managed in the impoundments.

The waste constituents dissolved in the leachate are transported via aqueous-phase migration through the unsaturated zone (soil layer under the WMU) to the underlying saturated zone (i.e., groundwater) and then downgradient to a hypothetical receptor. For this analysis, the exposure concentration was evaluated for the intake point of a hypothetical residential drinking water well (hereafter referred to as the “receptor well”). Figure 5-4 shows the conceptual model of the groundwater fate and transport of contaminant releases from a WMU to a downgradient receptor well. Further details about the modeled receptor well receptor are provided in Section 5.2.2.2.

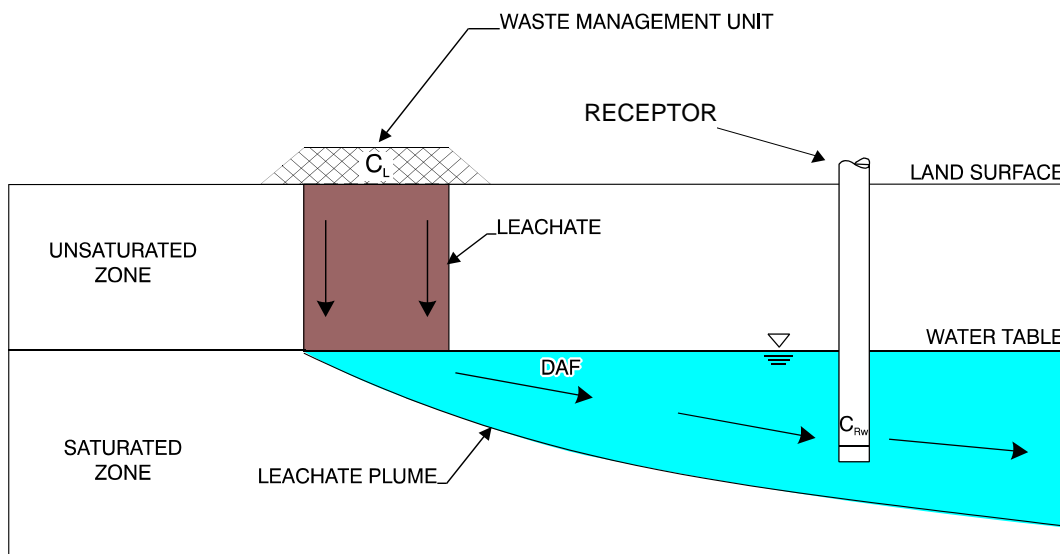


Figure 5-4. Schematic diagram of groundwater modeling scenario.

5.2.2.1 Groundwater Model. The transport of leachate from the WMU through the unsaturated and saturated zones is evaluated quantitatively using the EPACMTP (U.S. EPA, 2003a,b,c; 1997a). EPA used the EPACMTP model to make regulatory decisions for wastes managed in land disposal units (landfills, surface impoundments, waste piles, or land application units) for a number of EPA hazardous waste regulatory efforts. EPACMTP simulates flow and transport of contaminants in the unsaturated zone and aquifer beneath a waste disposal unit to

yield the concentration arriving at a specified receptor location. The source models described in Section 5.1 determine the leachate concentration that is used as input to the EPACMTP. New functionality was added to the EPACMTP model, as described in the addendum to the *EPACMTP Technical Background Document* (U.S. EPA, 2003c), to create the dynamic, mass-conservative linkage between the source models and the EPACMTP.

The groundwater model accounts for advection, hydrodynamic dispersion, equilibrium linear or nonlinear sorption, and transformation processes via chemical hydrolysis. In this analysis, data were compiled from the scientific literature to develop organic carbon partition coefficients for organic constituents. More information on organic constituent-specific partition coefficients is presented in Appendix C. Soil-water partition coefficients for metal constituents were derived from nonlinear empirical isotherms generated from the equilibrium speciation model MINTEQ2A, and their use is described in more detail in the following paragraphs.

EPACMTP is a composite model that consists of two coupled modules: (1) a one-dimensional (1-D) module that simulates infiltration and dissolved contaminant transport through the unsaturated zone, and (2) a saturated zone flow and transport module that includes three groundwater transport solution options: (1) fully 3-D transport; (2) quasi-3-D transport (a combination of cross-sectional and areal solutions); and (3) pseudo-3-D transport (hybrid analytical and numerical solution). The applicability and appropriateness of using each of the transport solution options depends on the problem considered. Using the quasi-3-D option instead of the fully-3-D option will result in a savings of CPU time and memory requirements. However, because the pseudo-3D solution is the most computationally efficient of the available solutions, using this solution option enables more Monte Carlo realizations to be included in the modeling analysis. Additionally, the pseudo-3D solution can accurately and efficiently generate full breakthrough curves at the receptor location(s). For these reasons, EPA chose to use the pseudo-3D solution option for this analysis.

The EPACMTP model algorithms are based on the assumption that the soil and aquifer are uniform porous media and that flow and transport are described by Darcy's law and the advection-dispersion equation, respectively. EPACMTP does not account for preferential pathways, such as fractures and macropores or facilitated transport, which may affect migration of strongly sorbing constituents, such as metals.

The primary transport mechanisms in the subsurface are downward movement along with infiltrating water flow in the unsaturated zone and movement along with ambient groundwater flow in the saturated zone. The advective movement in the unsaturated zone is 1-D, whereas the saturated zone module accounts for 3-D flow and transport. The model also considers mixing due to hydrodynamic dispersion in both the unsaturated and saturated zones. In the unsaturated zone, flow is gravity-driven and prevails in the vertically downward direction. Therefore, the flow is modeled in the unsaturated zone as 1-D in the vertical direction. It is also assumed that transverse dispersion (both mechanical dispersion and molecular diffusion) is negligible in the unsaturated zone. This assumption is based on the fact that lateral migration due to transverse dispersion is negligible compared with the horizontal dimensions of the WMUs. In addition, this assumption is environmentally protective because it allows the leading front of the constituent plume to arrive at the water table with greater peak concentration in the case of a finite source.

In the saturated zone, the movement of constituents is primarily driven by ambient groundwater flow, which in turn is controlled by a regional hydraulic gradient and hydraulic conductivity in the aquifer formation. The model takes into account the effects of infiltration from the waste source, as well as regional recharge into the aquifer. The effect of infiltration from the waste source is to increase the groundwater flow in the horizontal transverse and vertical directions underneath and in the immediate vicinity of the waste source, as may result from groundwater mounding. This 3-D flow pattern will enhance the horizontal and vertical spreading of the plume. Regional recharge outside of the waste source causes a (vertically) downward movement of the plume as it travels in the (longitudinally) downgradient groundwater flow direction. In addition to advective movement along with groundwater flow, the model simulates mixing of contaminants with groundwater due to hydrodynamic dispersion, which acts in the longitudinal direction (i.e., along the groundwater flow direction), as well as in horizontal and vertical transverse directions.

Leachate constituents can be subject to complex geochemical interactions in soil and groundwater, which can strongly affect their rate of transport in the subsurface. EPACMTP treats these interactions as equilibrium sorption processes. The equilibrium assumption means that the sorption process occurs instantaneously, or at least very quickly relative to the time scale of constituent transport. Although sorption, or the attachment of leachate constituents to solid soil or aquifer particles, may result from multiple chemical processes, EPACMTP lumps these processes together into an effective soil-water partition coefficient (K_d). The retardation factor (R) accounts for the effects of equilibrium sorption of dissolved constituents onto the solid phase. R , a function of the constituent-specific K_d and the soil or aquifer properties, is calculated as follows:

$$R = 1 + \frac{\rho_b \times K_d}{\phi} \quad (5-1)$$

where

R	=	retardation factor
ρ_b	=	soil or aquifer bulk density (mg)
K_d	=	solid-water partition coefficient (g/cm ³)
ϕ	=	water content (in unsaturated zone) or porosity (in saturated zone).

As modeled in the EPACMTP, the K_d of an organic constituent is assumed to be constant and is calculated as the product of the mass fraction of organic carbon in the soil or aquifer and a constituent-specific organic carbon partition coefficient (K_{oc}).

An isotherm is an expression of the equilibrium relationship between the aqueous concentration and the sorbed concentration of a metal (or other constituent) at a constant temperature. In the case of metals, EPACMTP accounts for more complex geochemical reactions by using effective sorption isotherms generated using EPA's geochemical equilibrium speciation model for dilute aqueous systems, MINTEQA2 (U.S. EPA, 1991b). The MINTEQA2 model was used to generate one set of isotherms for each metal, reflecting the range in geochemical environments expected at waste sites across the nation. The variability in geochemical environments was represented by five geochemical master variables (groundwater

composition, pH, concentration of iron oxide adsorption sites, concentration of anthropogenic organic acids in the leachate, and concentration of dissolved and particulate natural organic matter), and the MINTEQA2 modeling was repeated (separately for each metal) for numerous combinations of master variable settings. This procedure resulted in nonlinear K_d versus aqueous metal concentration curves for combinations of master variable settings spanning the range of reasonable values (U.S. EPA, 2003b).

For each metal, the resulting set of isotherms was tabulated into a supplementary input data file for use by the EPACMTP model. In the fate and transport modeling for a particular metal, EPACMTP is executed and the probability distributions for these five variables form the basis for the Monte Carlo selection of the appropriate adsorption isotherm. In modeling metals transport in the unsaturated zone, EPACMTP uses a range of K_d values from the nonlinear sorption isotherms. However, in modeling metals transport in the saturated zone, EPACMTP selects the lowest K_d from all available K_d values corresponding to concentrations less than or equal to the maximum water table concentration; for more details, see the *EPACMTP Technical Background Document* (U.S. EPA, 2003a). This simplification in the saturated zone is required for all solution options and is based on the assumption that after dilution of the leachate plume in groundwater, the concentrations of metals will typically be in a range where the isotherm is approximately linear. However, this assumption may not be valid when the metal concentrations in the leachate are exceedingly high. Although EPACMTP is able to account for the effect of the geochemical environment at a site on the mobility of metals, the model assumes that the geochemical environment at a site is constant and not affected by the presence of the leachate plume. In reality, the presence of a leachate plume may alter the ambient geochemical environment.

As can be seen in Equation 5-1, constituents with a low K_d (or K_{oc}) value will have an R close to 1.0, which indicates that they will move at nearly the same velocity as the groundwater. Constituents with high K_d values, such as certain semivolatile organic constituents and many metals, will have high R values and may move many times slower than groundwater.

The concentration of constituents in both the unsaturated zone and the saturated zone may be further attenuated by biochemical transformation reactions. The EPACMTP can, in principle, model a wide range of transformation processes so long as they can be lumped together as a first-order biodegradation process or a first-order chemical degradation process. In practice, a more conservative approach is adopted in which only the degradation of organic constituents via hydrolysis transformation processes is simulated. For those constituents that may be transformed via hydrolysis into toxic daughter products, EPACMTP can account for the formation and subsequent fate and transport of these daughter products. However, the COCs considered in this analysis are those identified in dye and pigment wastes and do not include significant degradation (or daughter) products.

Hydrolysis rates were calculated from chemical-specific hydrolysis rate constants using soil and aquifer temperature and pH values. Whereas chemical hydrolysis can be accurately modeled as a function of these factors, many other types of transformation processes, such as biodegradation, are much more site-specific and can be highly variable and, therefore, much more difficult to simulate using a generic model such as EPACMTP. EPA is therefore using a conservative assumption that these processes do not occur; that is, biodegradation is not

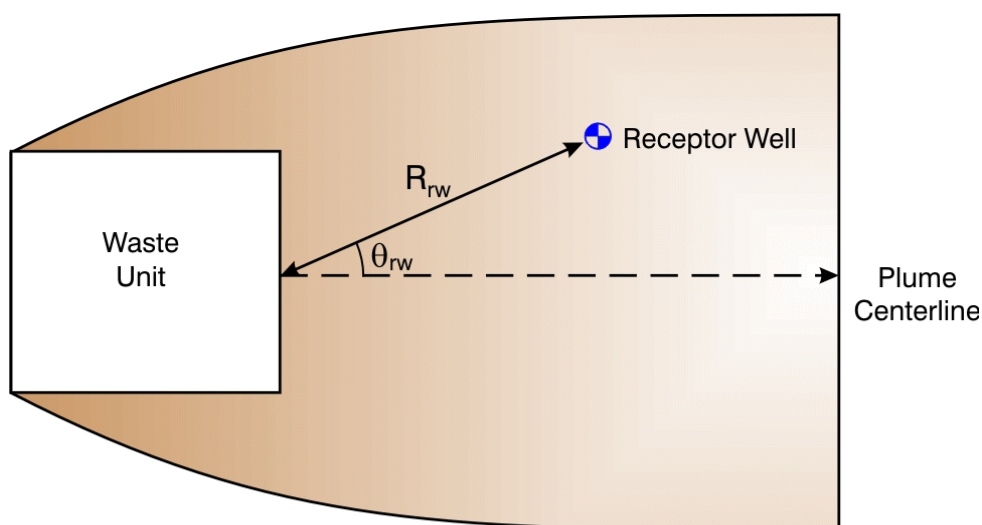
included. In the case of metals that may exist in multiple valence states, such as chromium (III) and chromium (VI), these species are modeled as separate constituents; no redox transformations are assumed to occur.

Whereas EPACMTP simulates steady-state flow in both the unsaturated zone and the saturated zone; contaminant transport can be modeled as either steady state or transient. Modeling flow as steady state assumes that the infiltration and recharge, as well as ambient groundwater flow conditions, are representative of long-term average conditions that do not change over the duration of the simulation process. The model does not account for transient flow conditions that may result from temporal (e.g., seasonal) variations in precipitation and regional groundwater flow.

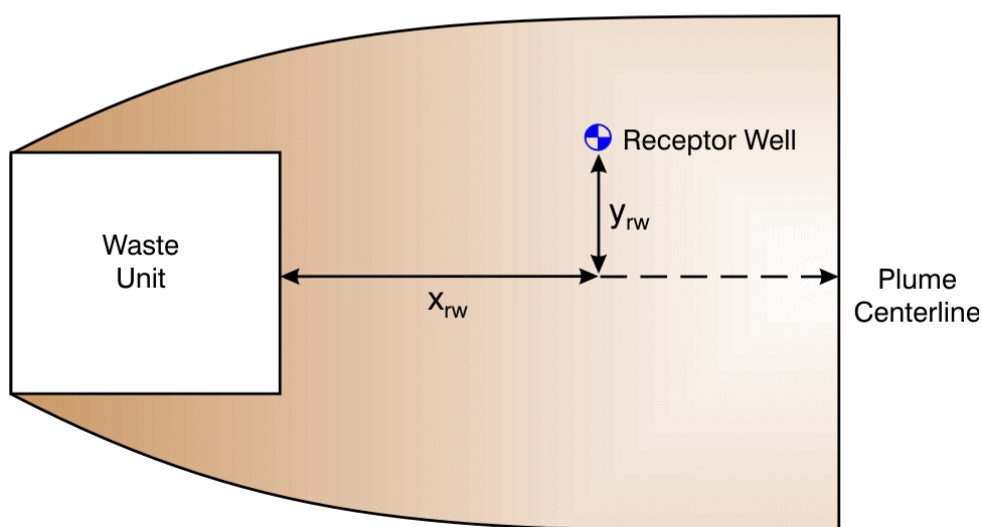
The EPACMTP model can accommodate either steady-state or transient contaminant transport. Steady-state transport modeling corresponds to a very conservative risk scenario in which a WMU continues to release contaminants indefinitely (continuous source), so that eventually the receptor well concentration reaches a constant, never-changing value. In this analysis, transient transport simulations were performed. This type of analysis is referred to as a “finite source” scenario because it explicitly takes into account the finite time period over which a WMU operates and releases contaminants, as well as the actual time it may take contaminants to reach the receptor well.

The output of EPACMTP is a prediction of the contaminant concentration arriving at a downgradient groundwater receptor location and is a time-dependent concentration, corresponding to the finite source scenario. In the latter case, the model can calculate either the peak concentration arriving at the well or a maximum time-averaged concentration. In this analysis, a maximum time-averaged concentration was calculated for all constituents based on the exposure duration for each receptor type. Because the subsurface migration of some highly sorbing constituents (e.g., lead) may be very slow, it may take a long time for the plume to reach the receptor well, and the maximum exposure may not occur until a very long time after the WMU ceases operations. This time delay may be on the order of thousands of years. In this analysis, a time period of 10,000 years following WMU closure was modeled. If the model predicts that the maximum exposure has not yet occurred after 10,000 years, the actual receptor concentration at 10,000 years is used in the risk calculations.

5.2.2.2 Groundwater Receptor. The EPACMTP can evaluate the exposure concentration at the intake point of a hypothetical residential drinking water well located at a specified distance from the downgradient edge of the WMU and at a specified depth below the water table. The location of the receptor well intake point is ultimately defined in terms of its x , y , and z coordinates relative to the center of the WMU’s downgradient edge (see Figure 5-5(b), in which x_{rw} refers to the downgradient distance from the edge of the WMU, y_{rw} refers to the horizontal transverse distance from the plume centerline; z_{rw} refers to the vertical distance below the water table). The planar coordinates, x_{rw} and y_{rw} , may be explicitly prescribed or derived from a radial distance, R_{rw} , and angle of rotation, θ_{rw} , off the plume centerline, as pictured in Figure 5-5(a). The shaded areas in Figures 5-5(a) and (b) conceptually represent the horizontal extent of the contaminant plume.



(a)



(b)



Region of Contaminant Plume

Figure 5-5. Schematic plan view showing procedure for determining the downstream location of the receptor well: (a) well location determined using radial distance, R_{rw} , and angle off center, θ_{rw} ; and (b) well location generated uniformly within plume limit.

In this analysis, the x_{rw} and y_{rw} distances are derived from a radius R_{rw} and angle θ_{rw} and are defined as

$$\begin{aligned} x_{rw} &= R_{rw} \cos(\theta_{rw}), \text{ and} \\ y_{rw} &= R_{rw} \sin(\theta_{rw}) \text{ or the distance from the plume centerline to the plume edge,} \\ &\quad \text{whichever is smaller.} \end{aligned}$$

The z_{rw} distance is defined as

$$z_{rw} = 0 \text{ to } 10 \text{ m (or to the base of aquifer if the aquifer thickness is less than } 10 \text{ m).}$$

The well is placed at a downgradient distance of up to 1 mi, based on a nationwide distribution of the nearest downgradient residential or municipal wells from a survey of Subtitle D municipal solid waste landfills (U.S. EPA, 1993). This distribution is used to represent R_{rw} and is presented in Appendix E. It is assumed that the same distribution holds for other types of WMUs, as well. A uniform distribution within the range of 0 to 90 degrees is used to model θ_{rw} . The limits on y_{rw} place the receptor well within the lateral extent of the plume (because of symmetry around the plume centerline, only one-half of the lateral extent of the plume is actually modeled). The constraints on the vertical depth of the well intake point reflect the fact that residential wells are expected to be in the shallower portion of the aquifer. The z location of the well is modeled using a uniform distribution with limits of zero (i.e., well at the water table) to 10 m or the total saturated aquifer thickness if the aquifer is less than 10 m thick.

Assuming a square WMU, so that the square root of the WMU area equals the length and width of the unit, the lateral extent of the plume, which is used to determine the y location of the well, is approximated as

$$\text{Plume edge} = 0.5\sqrt{AREA} + 3\sqrt{\alpha_T \times 2(\sqrt{AREA} + x_{rw})} \quad (5-2)$$

where

$$\begin{aligned} AREA &= \text{area of the WMU} \\ \alpha_T &= \text{horizontal transverse aquifer dispersivity} \\ x_{rw} &= \text{downgradient distance of receptor well from edge of the WMU.} \end{aligned}$$

The first term on the right side of this equation represents the contribution from the WMU area itself to the extent of the plume. The factor 0.5 represents the fact that the plume edge is calculated from the plume centerline (i.e., only one-half of the actual plume width is considered).

The second term on the right side represents the lateral spreading of the plume due to transverse dispersion, with increasing downgradient distance from the WMU. The magnitude of dispersive effects is expressed in the α_T parameter. The $AREA$ and x_{rw} terms represent the total downgradient plume travel distance, starting from the upgradient edge of the WMU.

The above approximation for the lateral extent of the contaminant plume is based on the assumption that plume spreading in the horizontal transverse direction is caused by dispersive mixing, which results in a Gaussian profile of the plume cross-section. Under this assumption, the approximation for the plume boundary will capture 99.7 percent of the contaminant mass. The above approximation, however, does not take into account advective spreading of the plume due to lateral groundwater flow in the vicinity of the WMU and may, therefore, underestimate the actual lateral extent of the plume. However, in this case, the approximation will be conservative, as it will prevent placement of the well at the outer fringes of the plume, which will have a lower concentration compared to a well that is closer to the plume centerline.

When EPACMTP was originally developed, the angle of the receptor well off of the plume centerline (see Figure 5.5(a)) was typically generated randomly, and x_{rw} and y_{rw} (see Figure 5.5(b)) were derived from R_{rw} and θ_{rw} . However, since that time, an improvement has been made to the modeling methodology that seeks to reduce the potential bias in observed concentrations associated with large WMU areas when the observation location is derived relative to the center of the downgradient edge of the WMU. This improved methodology, detailed in the *EPACMTP Technical Background Document* (U.S. EPA, 2003a), was used in the current analysis and allows the point of origin for the receptor location to move to the corner of the idealized WMU when the size of the WMU is large relative to R_{rw} . Regardless of the point of origin, the constraint on y_{rw} relative to the edge of the plume still applies.

For this assessment, the depth of the well was varied uniformly throughout the aquifer thickness or throughout the upper 10.0 m of the aquifer thickness, whichever is less. That is, the well depth is never allowed to exceed 10 m below the water table. This limitation for well depth has been used in previous listing determinations primarily for two reasons: (1) to be consistent with a residential well scenario (these wells are generally shallow because of the increased costs of drilling a deeper well) and (2) to produce a conservative estimate of risk (because the infiltration rate is generally lower than the groundwater seepage velocity, groundwater plumes generally tend to be relatively shallow).

5.2.2.3 Surface Water Receptor. The EPACMTP model was used to evaluate ecological risks due to exposure to surface water contaminated via incoming groundwater. Specifically, the model was used to determine the mass flux at a hypothetical interface between the saturated zone and a surface water body. In the model, the interface between groundwater and surface water is located downstream of the WMU along the plume centerline (see Figure 5-6). Some conservative simplifying assumptions have been made to successfully represent what is known to be a dynamic process between the groundwater and surface water within a Monte Carlo frame work, evaluate the risks due to exposure to contaminated surface water, and maintain computational efficiency. The primary simplifying assumptions are as follows:

1. The interface between groundwater and surface water is assumed to be perpendicular to the direction of regional groundwater flow,
2. The interface is infinite in its lateral extent so as to intercept the entire width of the dissolved contaminant plume,

3. The intercepting surface water body fully penetrates the saturated region of the subsurface, and
4. A fraction of the total mass in the plume intercepts the surface water body.

Contaminant mass fluxes generated by the pseudo-3D transport module of EPACMTP at the interface between groundwater and surface water are calculated from a time series of concentration values evaluated where the interface intersects the plume center line, as shown in Figure 5-6. The distance downstream of the WMU to this intersection is designated as x_{sw} (the EPACMTP parameter name for x_{sw} is DISSW, as presented in Appendix E). These transient point concentrations, $C^o(t, x_{sw})$, represent the mean dissolved concentration of a constituent inside a virtual footprint representing the WMU at a time t in years and distance x_{sw} in meters (see discussion of pseudo-3D transport module of EPACMTP in the Technical Background Document [U.S. EPA 2003a] for more details). $C^o(t, x_{sw})$, in mg/L, is convenient for calculating the total mass flux to the intercepting waterbody at time t and distance x_{sw} by using the following equation:

$$\dot{m}(t, x_{sw}) = C^o(t, x_{sw}) \times Y_S \times Z_S \times v \times 1000 \frac{L}{m^3} \times 0.001 \frac{g}{mg} \quad (5-3)$$

where

$\dot{m}(t, x_{sw})$	=	mass flux at time t and location x_{sw} (g/yr)
$C^o(t, x_{sw})$	=	mean dissolved constituent concentration within the virtual WMU footprint at time t and distance x_{sw} (mg/L)
Y_S	=	width of the virtual WMU footprint (m)
Z_S	=	depth of the virtual WMU footprint (m)
v	=	average groundwater velocity down stream of the WMU (m/yr).

EPACMTP reports the peak value of mass flux and its arrival time at the interface. To calculate the surface water concentration, the peak mass flux value is divided by a volumetric water flux (15 ft³/s) representing the flow rate in a standard third-order stream intersecting the contaminant plume (Keup, 1985).

Assumption 3 is a conservative simplification for numerical and computational convenience that may result in an overly conservative estimate of the mass flux, because clearly the entire mass from the groundwater plume will not reach the surface water body. Thus, it was assumed that some fraction of the total mass intercepts the surface waterbody. Consistent with the Monte Carlo framework for the analysis, this fraction was randomly sampled between a value of zero and one to take into consideration all possible scenarios. The resulting equation for estimating the surface water concentration is as follows:

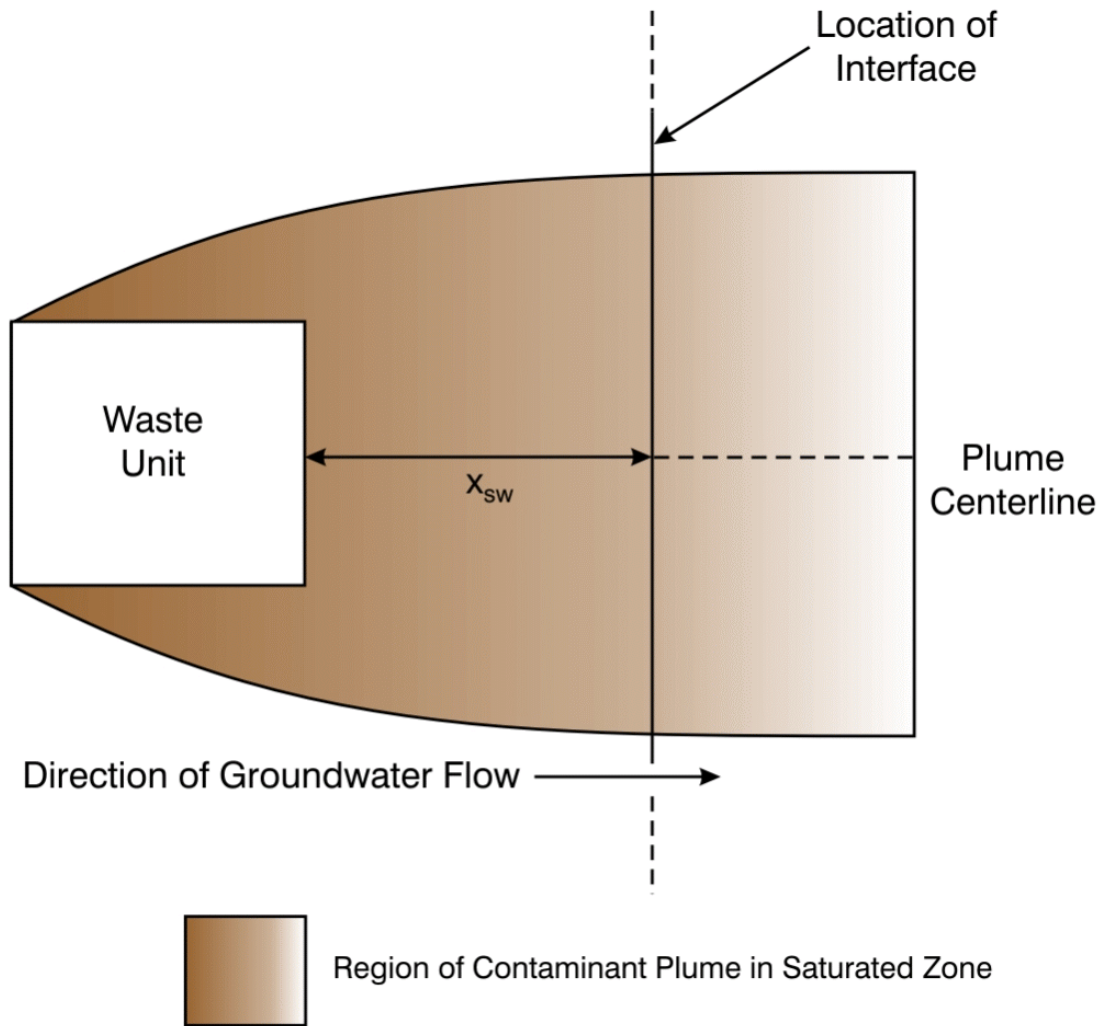


Figure 5-6. Schematic plan view showing the conceptualized interface between groundwater and surface water.

$$C_{SW}^{Peak} = \frac{\dot{m}^{Peak}}{Q} \times f_{intercept} \quad (5-4)$$

where

C_{SW}^{Peak}	=	peak concentration at location x_{sw} (g/yr)
\dot{m}^{Peak}	=	peak mass flux at location x_{sw} (g/yr)
Q	=	volumetric water flux in the surface water body (m ³ /yr)
$f_{intercept}$	=	fraction of interception [U(0,1)].

Monte Carlo simulations to support this assessment have been conducted for both the groundwater exposure pathway and the groundwater-to-surface-water pathway. The screening calculation for surface waters requires a representative peak surface water concentration derived from mass fluxes across the groundwater-surface water interface. The back-calculation approach for determining the maximum allowable waste concentration for organic constituents makes use of the proportionality that exists between the leachate concentration and observed concentrations at receptor locations, as described in Section 9.1. In the case of metals, however, groundwater concentrations predicted by EPACMTP, using empirical MINTEQA2 isotherms, cannot be assumed to change linearly with changes in the leachate concentration. Given that the source models treat metals transport as a linear process, the maximum allowable waste concentration is determined with a backward calculation across the source module so long as the 90th percentile peak surface water concentration is equal to the constituent's ambient water quality criteria (AWQC) (see Section 9.2 discussion on receptor wells).

5.2.3 Shower Pathway

Constituents that have reached a residential groundwater well may be extracted and used for domestic purposes (e.g., showering). In this analysis, it was assumed that constituents that reach a residential groundwater well would be used not only as drinking water, but also for showering. This type of exposure is called indirect exposure because a constituent is passed indirectly from contaminated media to a receptor. The indirect exposure from showering occurs when vapors are generated as a result of volatilization of the constituent in the domestic water drawn from the groundwater well.

Exposure to constituents released from showering activity was evaluated for the adult resident only. Young children are assumed to take primarily baths and thus not to encounter significant exposure to constituents released during showering activity. The adult resident is assumed to take one shower per day. The duration of each shower event is assumed to be a distribution of values ranging from 1 to 60 minutes. Exposure due to showering with contaminated residential groundwater is assumed to occur 350 days of the year, which is consistent with exposure frequency assumptions made for other exposure pathways.

Appendix G presents the algorithms that are the basis of the shower model and the exposure factors used in the shower model.

A primary limitation of the shower model is that it only calculates the gas-phase concentration of a constituent resulting from showering activity. Although previous versions of the model estimated emissions and air concentrations resulting from other household uses of water, these sources are not included in this version because of the relatively low risk from exposure to non-showering sources and because the input data needed to estimate household chemical concentrations from other sources are not available. Section 5.2.3.1 describes these and other uncertainties and limitations in greater detail.

5.2.3.1 Model Description. The shower model estimates the change in the shower air concentrations over time based on the mass of constituent lost by the water and the air exchange rate between the shower, the bathroom, and the rest of the house. The shower model used for this analysis is based on differential equations presented in McKone (1987) and Little (1992a). The differential equations were solved using finite difference numerical integration.

The shower model estimates the concentration of a constituent in indoor air that results from the mass transfer of the constituent from water to shower air. The following equation estimates the overall mass transfer coefficient from tapwater to air from showering:

$$K_{ol} = \beta \times \left(\frac{2.5}{D_w^{2/3}} + \frac{1}{D_a^{2/3} H'} \right)^{-1} \quad (5-5)$$

where

K_{ol} = overall mass transfer coefficient (cm/s)

β = proportionality constant (cm-s)^{-1/3}

D_w = diffusion coefficient in water (cm²/s) (contaminant-specific, provided in Attachment B)

D_a = diffusion coefficient in air (cm²/s) (contaminant-specific, provided in Attachment B)

H' = dimensionless Henry's law constant (calculated using the equation in Appendix G).

Equation 5-5 was presented in McKone (1987) with the proportionality constant noted as a dimensionless value. In a correspondence to the editor, Little (1992b) indicated that β is not dimensionless. The correct units are noted above. Because McKone did not provide an estimate for the proportionality constant, a value was derived using data for benzene. The modeled results using the benzene-derived value (216 (cm-s)^{-1/3}) compared favorably to experimental data for several organic compounds of varying volatility (Coburn, 1996).

The constituent emission rate was estimated from the change in the shower water concentration as the water falls, which is calculated using the overall mass transfer coefficient as follows:

$$dc/dt = - K_{ol}(A/V)(c - y_s/H') \quad (5-6)$$

where

c = liquid phase (droplet) constituent concentration (µg/cm³ or mg/L)

t = time (s)

K_{ol} = overall mass transfer coefficient (cm/s)

A = total surface area for mass transfer (cm²)

V = total volume of water within the shower compartment (cm³)

y_s = gas-phase constituent concentration in the shower (µg/cm³ or mg/L)

H' = dimensionless Henry's law constant (calculated using the equation in Appendix G).

Consequently, in addition to the overall mass transfer coefficient, the emission rate of a contaminant within the shower is dependent on the surface-area-to-volume ratio of the shower water and the concentration driving force between the water and the shower air. Equation 5-6 above is equivalent to Equation 4 presented in Little (1992a).

The shower emissions were modeled based on falling droplets as a means of estimating the surface-area-to-volume ratio for mass transfer and the residence time of the water in the shower compartment. Equation 5-6 was integrated assuming the constituent concentration in the gas phase is constant over the timeframe of the droplet fall. The time required for a droplet to fall equals the distance traveled (nozzle height) divided by the water droplet velocity. The ratio of the surface area to volume for the droplet equals $6/d_p$ (assuming a spherical shape where d is droplet diameter). Given the assumption that the drops fall at terminal velocity, the surface-area-to-volume ratio and the residence time were determined based solely on droplet size. A droplet size of approximately 1 mm (0.098 cm) was selected. The terminal velocity for the selected droplet size is approximately 400 cm/s. The fraction of constituent emitted from a water droplet at any given time was calculated by integrating Equation 5-6 and rearranging as follows:

$$f_{em,s} = 1 - C_{out}/C_{in} = (1 - f_{sat})(1 - e^{-N}) \quad (5-7)$$

where

$f_{em,s}$ = fraction of constituent emitted from the shower droplet (dimensionless)

C_{out} = droplet constituent concentration at shower floor/drain (mg/L)

C_{in} = droplet constituent concentration entering the shower (mg/L)

f_{sat} = fraction of gas-phase saturation (dimensionless) (calculated using the equation in Appendix G)

N = dimensionless overall mass transfer coefficient (calculated using the equation in Appendix G).

d_p = droplet diameter = 0.098 (cm)

v_t = terminal velocity of droplet = 400 (cm/s)

h = nozzle height (cm).

Equation 5-7 is equivalent to the equation presented in Little (1992a, Equation 5) for liquid-phase concentration change, except that Little (1992a) used the total surface area for mass transfer (lumped with K_{ol}) and the shower flow rate to calculate an overall mass transfer coefficient with dimension, where $N = (K_{ol} \times A)/Q_L$ and Q_L is the volumetric flow rate of the water. This equation can be alternately presented as $N = (K_{ol})(A/V)(DropRT)$, where $DropRT$ represents the time it takes for a droplet to fall. The area-to-volume ratio (A/V) is expressed in Equation 5-7 as $6/d_p$. The drop residence time was calculated using the height and the velocity of the droplet (h/v_t) (see Appendix G). Because the surface area of the droplet was a difficult

value to obtain, the above approach was developed so that the total surface area for mass transfer was not needed.

After calculating the fraction of constituent emitted from the shower water, the model converts the fraction to a mass emitted for each time step. The mass was calculated by multiplying the total mass entering the shower during a given time step by the fraction emitted. The total mass entering the shower was determined by multiplying the concentration in the shower water by the flow rate of the shower head and the amount of the time step. A time step of 0.2 minutes was used in the model. The gas-phase constituent concentration in the shower was calculated for each time step for the duration of the shower. The air exchange rate between the shower and the bathroom was included in the estimation of the gas-phase concentration of the constituents in the shower.

$$y_{s, t+ts} = y_{s, t} + \frac{\left[E - (Q_{sb} \times (y_{s, t} - y_{b, t}) \times ts) \right]}{V_s \times 1000} \quad (5-8)$$

where

- $y_{s, t+ts}$ = gas-phase constituent concentration in the shower at the end of time step (mg/L)
- $y_{s, t}$ = gas-phase constituent concentration in the shower at the beginning of time step (mg/L)
- Q_{sb} = volumetric gas exchange rate between shower and bathroom (L/min)
- $y_{b, t}$ = gas-phase constituent concentration in the bathroom at the beginning of time step (mg/L)
- ts = time step (min) = 0.2
- E = mass of constituent emitted from shower during time step (mg)
- V_s = volume of shower stall (m^3)
- 1000 = conversion factor (L/m^3).

The shower model also provides direct estimates of bathroom exposure. The gas-phase constituent concentration in the bathroom was estimated using Equation 5-9 for each time step of the exposure duration.

$$y_{b, t+ts} = y_{b, t} + \frac{\left[(Q_{sb} \times (y_{s, t+ts} - y_{b, t}) - Q_{bh} \times (y_{b, t} - y_{h, t})) \right] \times ts}{V_b \times 1000} \quad (5-9)$$

where

$y_{b, t+ts}$ = gas-phase constituent concentration in the bathroom at end of time step (mg/L)

$y_{b, t}$ = gas-phase constituent concentration in the bathroom at beginning of time step (mg/L)

Q_{bh} = volumetric gas exchange rate between bathroom and house (L/min)

Q_{sb} = volumetric gas exchange rate between shower and bathroom (L/min)

$y_{s, t+ts}$ = gas-phase constituent concentration in the shower at end of time step (mg/L)

$y_{h, t}$ = gas-phase constituent concentration in the house at beginning of time step (mg/L)

ts = time step (min) = 0.2

V_b = volume of bathroom (m^3)

1000 = conversion factor (L/m^3).

The gas-phase constituent concentration in the house was assumed to be negligible in comparison with the gas-phase constituent concentration in the bathroom. Therefore, the gas-phase concentration of the constituent in the house, $y_{h, r}$, was set to zero.

The average air concentrations in the shower and bathroom were calculated by averaging the concentrations from each time step over the duration of the shower and bathroom use. From the average shower concentration and the average bathroom concentration, a time-weighted average concentration for an entire day was calculated. This time-weighted concentration was used to estimate chronic health impacts due to inhalation exposures from showering.

5.2.3.2 Model Limitations and Uncertainties. A primary limitation of this model is that the gas-phase concentration of a constituent in household air results solely from showering activity. Receptors are exposed to emissions via inhalation for time spent in the shower while showering, in the shower stall after showering, and in the bathroom after showering. Previous versions of the shower model included emissions due to other household uses of water and risks due to inhalation for time spent in the remainder of the house. However, the risk from inhalation exposures in the remainder of the house was shown to be several orders of magnitude lower than the risk from inhalation exposures in the bathroom and during showering (U.S. EPA, 1997b). In addition, there are few data available to estimate the input parameters needed to calculate exposure concentrations from other household activities, including variables such as house volume, air exchange rate between the house and outside air, and exposure time in the house. Given the low risk due to exposure from the remainder of the house and the lack of available data to estimate house contaminant concentrations, the current version of the shower model was designed to focus on showering as the greatest source of inhalation exposure and risk due to use of contaminated water.

The input parameter values are another source of uncertainty for the shower model. To select values for the shower properties (shower and bathroom volume, nozzle height, and flow rate), mid-range values that were reported in the literature were generally used. Although fixing shower parameters does not capture variability in these inputs when the model is run using these fixed values, the results compare favorably to experimental data for numerous organic compounds of varying volatility (Coburn, 1996). The droplet properties (diameter and velocity) are also constants, with their values based on a correlation to existing data. The largest uncertainty is likely in the volumetric exchange rates used between the shower and bathroom and the bathroom and the rest of house. These values, 300 L/min for the exchange rate between the bathroom and house, and 100 L/min for the exchange rate between the shower and bathroom, were derived from McKone (1987). The range of values reported in a five-study summary (Little, 1992a), however, showed ranges of 35 to 460 L/min for the exchange between the shower and bathroom and 38 to 480 L/min for the exchange between the bathroom and the rest of the house. Such a large range of volumetric exchange rates is an uncertainty in the shower model's estimation of contaminant concentrations.

5.3 References

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6.0 Human Exposure Assessment

This section describes the human exposure assessment that was conducted for the dyes and pigments listing risk assessment to determine or estimate the magnitude, frequency, duration, and route of exposure to dye and pigment contaminants that an individual may experience. The term “exposure,” as defined by the EPA exposure guidelines (U.S. EPA, 1992), is the condition that occurs when a contaminant comes into contact with the outer boundary of the body. The exposure of an individual to a contaminant completes an exposure pathway (i.e., the course a constituent takes from the WMU to an exposed individual). Once the body is exposed, the constituent can cross the outer boundary and enter the body. The amount of contaminant that crosses and is available for adsorption at internal exchange boundaries is referred to as the “dose” (U.S. EPA, 1992).

The dyes and pigments listing risk assessment evaluated the risk from dye and pigment contaminants to receptors in the vicinity of a WMU. The individuals evaluated were those residents closest to the WMU. The distances from the WMU to the residents were taken from empirical data and are described in detail in Section 4.3. Section 6.1 presents an overview of the receptors and selected exposure pathways considered for this assessment. Section 6.2 presents exposure factors (i.e., values needed to calculate human exposure) used in the analysis. Section 6.3 describes the methods used to estimate dose, including average daily dose (ADD) and lifetime average daily dose (LADD).

6.1 Receptors and Exposure Pathways

Table 6-1 lists each receptor along with the specific exposure pathways that apply to that receptor. The groundwater pathways were evaluated separately from other pathways because exposures may occur on a different timeframe from that of other pathways. Air emissions occur immediately after waste disposal, whereas it may take hundreds of years for most contaminants to leach from a WMU, travel in the groundwater, and reach a residential dwelling. As indicated in Table 6-1, not all receptors are exposed through the same pathways. Adult and child residents are exposed via inhalation of air and ingestion of groundwater. Only adult residents are exposed to the groundwater pathway via the inhalation of vapors generated from constituents volatilizing from well water during showering.

6.1.1 Childhood Exposure

Children are an important subpopulation to consider in a risk assessment because they are likely to be more susceptible to exposures than adults; compared with adults, children may drink more fluids per unit of body weight. This higher intake-rate-to-body-weight ratio can result in a higher ADD than adults experience.

Table 6-1. Receptors and Exposure Pathways

Receptor	Inhalation of Ambient Air	Ingestion of Drinking Water	Inhalation of Indoor Air (Shower) ^b
Adult resident	✓		
Child resident	✓		
Adult resident ^a		✓	✓
Child resident ^a		✓	

^a Groundwater pathways were considered separately for the adult resident and the child resident because the timeframe for groundwater exposure often is not consistent with that of other exposure pathways.

^b Showering is explicitly considered for adults and not for children; however, the results can be considered to bracket the risk for this pathway. In terms of noncancer, the hazard quotient is calculated based on the air concentrations, which are the same regardless of age, and the time spent in the shower, which is based on data for all ages. For cancer, adults are typically the most sensitive receptor due to the longer-term exposure.

As children mature, their physical characteristics and behavior patterns change. To capture these changes in the analysis, the life of a child was considered in stages represented by the following cohorts: cohort 1 (ages 1 to 5), cohort 2 (ages 6 to 11), cohort 3 (ages 12 to 19), and cohort 4 (ages 20 to 70). Associated with each cohort are distributions of values, called “exposure parameters,” that are required to calculate exposure to an individual. The exposure parameter distributions for each cohort reflect the physical characteristics and behavior patterns of that age range. Data from the *Exposure Factors Handbook* (EFH) were used to derive distributions appropriate for each cohort (U.S. EPA, 1997a,b,c). The distributions for the 20- to 70-year-old cohort were the same as those used for adult receptors.

The development of the child exposure parameters consisted of two steps:

1. Define the start age of the child
2. Select the exposure duration of the child.

To capture the higher intake-rate-to-body-weight ratio of children, a start age of 1 was selected. The distribution of exposure duration for cohort 1 (a 1- to 5-year-old) was used to define exposure duration for each of the 10,000 iterations in the probabilistic analysis.

6.1.2 Exposure Pathways

Human receptors may come into contact with COCs present in environmental media through a variety of pathways. The exposure pathways considered in this assessment were inhalation of ambient air, inhalation of indoor air vapors from contaminated groundwater via showering, and ingestion of drinking water from contaminated groundwater sources. As noted previously, exposure to groundwater was considered separately from other pathways in this analysis because the time scales differ from those for other pathways and the groundwater plume occurs in a limited area relative to potential receptor locations. Furthermore, aboveground

receptors are randomly located along the centerline of an atmospheric plume whose direction is determined by surface wind patterns, and they do not necessarily coincide with the location of the groundwater plume.

6.1.2.1 Inhalation of Ambient Air. Both vapors and particles can be inhaled in ambient air by a receptor, but as explained in Section 5, only vapors were considered in this analysis. Adult and child receptors are affected via direct inhalation.

6.1.2.2 Inhalation of Indoor Air. Groundwater is contaminated from COCs leaching from the WMU into the vadose zone and migrating into the groundwater. Groundwater extracted from an offsite well was assumed to be used in the residence for showering by the adult residential receptor. COCs can volatilize from shower water, thus resulting in inhalation exposures.

6.1.2.3 Ingestion of Drinking Water. In addition to inhalation of indoor air, ingestion of drinking water (also an indirect groundwater pathway) was considered in the analysis. Groundwater from the offsite well was assumed to be used as a source of drinking water for residents (adult and child). Both groundwater pathways (i.e., inhalation of indoor shower air and ingestion of drinking water) were considered separately from other pathways because they typically occur on a different timeframe and receptor locations do not necessarily coincide.

6.2 Exposure Factors

The exposure factors used in this risk assessment are listed in Table 6-2, along with the data sources and their representation were used as either a distribution or a fixed value in the Monte Carlo analysis. These exposure factors were used to calculate the dose of a chemical based on contact with contaminated media or food, the duration of that contact, and the body weight of the exposed individuals. The primary data source of human exposure model inputs used in this risk assessment was EPA's EFH (U.S. EPA, 1997a,b,c). The EFH summarizes data on human behaviors and characteristics related to human exposure from relevant key studies and provides recommendations and associated confidence estimates on the values of exposure factors. These data were carefully reviewed and evaluated for quality before being included in the EFH. EPA's evaluation criteria included peer review, reproducibility, pertinence to the United States, currency, adequacy of the data collection period, validity of the approach, representativeness of the population, characterization of the variability, lack of bias in study design, and measurement error (U.S. EPA, 1997a,b,c).

For probabilistic risk analyses, probability distribution functions (PDFs) are developed from the values in the EFH. Appendix H presents the exposure factors used in the dyes and pigments assessment probabilistic risk analysis. Appendix H also describes the rationale and data used to select the parametric models (gamma, lognormal, and Weibull) for those exposure factors that were varied in the probabilistic analysis.

Table 6-2. Human Exposure Factor Input Parameters and Data Sources

Parameter	Variable Type	Data Source
Body weight (adult, child)	Distribution	U.S. EPA (1997a)
Inhalation rate (adult, child)	Distribution	U.S. EPA (1997a)
Ingestion rate: drinking water (adult, child)	Distribution	U.S. EPA (1997a)
Shower contact time (adult)	Distribution	U.S. EPA (1997c)
Exposure duration (adult, child)	Distribution	U.S. EPA (1997c)
Exposure frequency (adult, child)	Fixed (constant)	U.S. EPA policy
Fraction contaminated: drinking water	Fixed (constant)	U.S. EPA policy
Event frequency: showering	Fixed (constant)	U.S. EPA (1997c)
Human lifetime (used in carcinogenic risk calculation)	Fixed (constant)	U.S. EPA policy

6.2.1 Intake Factors

6.2.1.1 Drinking Water Ingestion Rates. Use of groundwater from a contaminated well downgradient from a WMU is a pathway common to adult and child residents. Groundwater pathways were considered separately from other pathways for the adult resident and the child resident because the timeframe for groundwater exposure often is not consistent with that of other exposure pathways. Furthermore, aboveground receptors are randomly located and do not necessarily coincide with the location of the groundwater plume.

6.2.1.2 Inhalation Rates. The EFH reports inhalation values by age, gender, and activity pattern and for outdoor workers; however, it does not provide high-end values in most cases.

6.2.2 Other Exposure Factors

6.2.2.1 Body Weights. Distributions of body weight were developed for adult and child receptors based on data from the EFH.

6.2.2.2 Exposure Duration. Exposure duration refers to the amount of time that a receptor is exposed to a contaminant source. For this risk analysis, exposure duration was assumed to correspond with the receptor's residence time in the same house. Exposure durations were determined using data on residential occupancy from the EFH (U.S. EPA, 1997c). The data used to develop parameter information for resident receptors were age-specific. Thus, separate distributions were developed for adult and child residents. Exposure durations used in this risk assessment are provided in Appendix H.

6.2.2.3 Exposure Frequency. Exposure frequency is the frequency at which the receptor is exposed to the contaminated source during the exposure duration. Exposure

frequency is not expected to vary much, so distributions were not developed for exposure frequency. All receptors were assumed to be exposed to the contaminant source 350 d/yr. This value is based on an assumption that individuals are away from their homes (e.g., on vacation) approximately 2 weeks out of the year.

6.2.2.4 Lifetime and Averaging Time. Averaging time is the period of time over which a receptor's dose is averaged. To evaluate carcinogens, total dose was averaged over the lifetime of the individual, assumed to be 70 years. To evaluate noncarcinogens, dose was averaged over the last year of exposure because noncancer effects may become evident during less-than-lifetime exposure durations if toxic thresholds are exceeded. Essentially, this amounts to setting exposure duration and averaging time equal (i.e., both are equal to 1.0) so that they cancel each other in the equation for ADD. Thus, neither exposure duration nor averaging time is included in the ADD equation.

6.3 Dose Estimates

The purpose of the exposure assessment were to estimate the dose to each receptor by combining intake values with media concentrations. Estimates of exposure were based on the potential dose (e.g., the dose ingested or inhaled) rather than the applied dose (e.g., the dose delivered to the gastrointestinal tract) or the internal dose (e.g., the dose delivered to the target organ). This is generally consistent with the exposure metric used in most epidemiologic and toxicologic studies that serve as the basis for establishing the toxicological benchmarks used for risk assessment (see Section 7.0).

Doses from groundwater ingestion were calculated by multiplying the contaminant concentration with the respective intake rate on a per kilogram body weight basis. Doses were then summed over the period of time in which exposure occurred, resulting in an ADD received from ingestion exposure. The ADD was used to assess noncancer risk from ingestion exposures. Inhalation noncancer risks were evaluated using ambient air concentrations. For cancer effects, where the biological response is described in terms of lifetime probabilities even though exposure may not occur over the entire lifetime, dose is presented as a LADD. The LADD was used to assess cancer risks from each exposure route (i.e., inhalation and ingestion).

6.3.1 Average Daily Dose

For the purposes of this risk analysis, the ADD was defined as

$$ADD = C \times IR \quad (6-1)$$

where

- C = average concentration (mass/volume or mass/mass)
- IR = intake rate (mass/body weight mass/day, or volume/body weight mass/day).

Contaminant concentration represents the concentration of a chemical in a medium that contacts the body. The ADD was calculated over the exposure duration for each receptor.

6.3.2 Lifetime Average Daily Dose

The LADD used for assessing risks for carcinogenic effects was defined as

$$\text{LADD} = \frac{C \times \text{IR} \times \text{ED} \times \text{EF}}{\text{AT} \times 365} \quad (6-2)$$

where

C	=	average concentration (mass/mass or mass/volume)
IR	=	intake rate (mass/body weight mass/day, or volume/body weight mass/day)
ED	=	exposure duration (yr)
EF	=	exposure frequency (d/yr)
AT	=	averaging time (yr)
365	=	units conversion factor (d/yr).

As with the ADD, contaminant concentration represents the concentration of a chemical in a medium that contacts the body. Intake rate depends on the route of exposure; for example, it might be an inhalation rate or an ingestion rate. Exposure frequency is the number of days per year the receptor is exposed to the contaminated source during the exposure duration.

For cancer effects, biological responses are described in terms of lifetime probabilities, even though exposure may not be lifelong; consequently, the exposure duration (the length of time of contact with a contaminant) was used to average the ADD over a lifetime (70 years). The media concentrations used in the analysis were averaged over the duration of exposure.

6.4 References

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- U.S. EPA (Environmental Protection Agency). 1997c. *Exposure Factors Handbook, Volume III, Activity Factors*. EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC. August.

7.0 Human Health Toxicity Assessment

To characterize the risk from human exposures to the dye and pigment COCs, toxicity information on each COC was integrated with the results of the exposure assessment. A chemical's ability to cause an adverse health effect depends on the toxicity of the chemical, the chemical's route of exposure to an individual (either through ingestion or inhalation), the duration of exposure, and the dose received (the amount that a human ingests or inhales). For a risk assessment, the toxicity of a constituent is defined by a human health benchmark for each route of exposure. Essentially, a benchmark is a quantitative value used to predict a chemical's possible toxicity and ability to induce a health effect at certain levels of exposure. These health benchmarks are derived from toxicity data based on animal studies or human epidemiological studies. Each benchmark represents a dose-response estimate that relates the likelihood and severity of adverse health effects to exposure and dose. Because individual chemicals cause different health effects at different doses, benchmarks are chemical-specific.

Human health benchmarks for chronic oral and inhalation exposures were needed to conduct the risk characterization. This section presents the noncancer and cancer benchmarks used to evaluate human health effects that may result from exposure to constituents modeled for this risk assessment. Section 7.1 discusses human health benchmarks and their sources. Section 7.2 provides the benchmark values used in the analysis for each constituent.

7.1 Types and Sources of Health Benchmarks

Chronic human health benchmarks were used in this risk assessment to evaluate potential noncancer and cancer risks. Reference doses (RfDs) and reference concentrations (RfCs) were used to evaluate noncancer risk from oral and inhalation exposures, respectively, and oral cancer slope factors (CSFs), inhalation unit risk factors (URFs), and inhalation CSFs to evaluate risk for carcinogens. The benchmarks are chemical-specific and do not vary between age groups.

The RfD and RfC are estimates (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious noncancer effects during a lifetime. It is not a direct estimator of risk but rather a reference point to gauge the potential effects. At exposures increasingly greater than the RfD (or RfC), the potential for adverse health effects increases. Lifetime exposure above the RfD (or RfC), however, does not imply that an adverse health effect would necessarily occur (U.S. EPA, 2003).

The RfD and RfC are the primary benchmarks used to evaluate noncarcinogenic hazards posed by environmental exposures to chemicals. They are based on the "threshold" approach, which is the theory that there is a "safe" exposure level (i.e., a threshold) that must be exceeded

before an adverse noncancer effect occurs. RfDs and RfCs do not provide true dose-response information in that they are estimates of an exposure level or concentration that is believed to be below the threshold level or no observed adverse effects level (NOAEL). The degree of uncertainty and confidence levels in RfDs varies and is based on different toxic effects.

The CSF is an upper-bound estimate (approximating a 95 percent confidence limit) of the increased human cancer risk from a lifetime exposure to an agent. This estimate is usually expressed in units of proportion (of a population) affected per milligram of agent per kilogram body weight per day (mg/kg-d). The unit risk, which is calculated from the slope factor, is the upper-bound excess lifetime cancer risk estimated to result from continuous exposure to an agent at a concentration of 1 µg/L in water or 1 µg/m³ in air. That is, if unit risk = 1.5×10^{-6} µg/L, then 1.5 excess tumors are expected to develop per 1,000,000 people if exposed daily for a lifetime to 1 µg of the chemical in 1 L of drinking water (U.S. EPA, 2003). Unlike RfDs and RfCs, CSFs and URFs do not represent “safe” exposure levels; rather, they relate levels of exposure with a probability of effect or risk.

Several sources of human health benchmarks are available. Health benchmarks developed by the U.S. EPA were used whenever they were available. Human health benchmarks were obtained from these sources in the following order of preference:

1. Integrated Risk Information System (IRIS)
2. Superfund Technical Support Center Provisional Benchmarks
3. Health Effects Assessment Summary Tables (HEAST)
4. Various other EPA health benchmark sources
5. Agency for Toxic Substances and Disease Registry (ATSDR) minimal risk levels (MRLs)
6. California Environmental Protection Agency (CalEPA) chronic inhalation reference exposure levels (RELs) and cancer potency factors.

Integrated Risk Information System (IRIS)

Benchmarks in IRIS are prepared and maintained by EPA. IRIS is EPA’s electronic database containing consensus scientific positions on potential adverse human health effects that may occur from chronic exposure to environmental contaminants (U.S. EPA, 2003). Each chemical file contains descriptive and quantitative information on potential health effects. Health benchmarks for chronic noncarcinogenic health effects include RfDs and RfCs. Cancer classifications, oral CSFs, and inhalation URFs are included for carcinogenic effects. IRIS is the official repository of Agency-wide consensus information on human health risk.

To assess less-than-lifetime cancer risks (e.g., child) and address population variability (e.g., body weight differences among adults), inhalation CSFs were used in this risk assessment. By using inhalation CSFs, one may account for age-specific differences and population

variability in inhalation rate and body weight, as well as exposure duration and frequency. Inhalation URFs are not dependent on exposure factors (e.g., inhalation rate) and therefore cannot be used to address population variability or age-specific differences in exposure scenarios. Inhalation CSFs are not available from IRIS, so they were calculated for use in this risk assessment based on inhalation URFs (which are available from IRIS) using the following equation:

$$\text{inh CSF} = \text{inh URF} \times 70 \text{ kg} \div 20 \text{ m}^3/\text{d} \times 1,000 \text{ }\mu\text{g}/\text{mg} \quad (7-1)$$

In this equation, 70 kg represents average body weight; 20 m³/d represents average inhalation rate; and 1000 µg/mg is a units conversion factor (U.S. EPA, 1997). These standard estimates of body weight and inhalation rate were used in the calculation of the URF and were therefore used to calculate inhalation CSFs.

Superfund Provisional Benchmarks

The Superfund Technical Support Center (EPA's National Center for Environmental Assessment [NCEA]) derives provisional RfCs, RfDs, and CSFs for certain chemicals. These provisional health benchmarks can be found in Risk Assessment Issue Papers. Some of the provisional values have been externally peer reviewed. These provisional values have not undergone EPA's formal review process for finalizing benchmarks and do not represent Agency-wide consensus information.

Health Effects Assessment Summary Tables (HEAST)

HEAST is a listing of provisional noncarcinogenic and carcinogenic health benchmarks (RfDs, RfCs, CSFs, and URFs) derived by EPA (U.S. EPA, 1997). HEAST benchmarks are considered secondary to those contained in IRIS. Although the health benchmarks in HEAST have undergone review and have the concurrence of individual EPA program offices, either they have not been reviewed as extensively as those in IRIS or their data set is not complete enough to be listed in IRIS. HEAST benchmarks have not been updated in several years and are not recognized as Agency-wide consensus information.

Other EPA Health Benchmarks

EPA has also derived provisional health benchmark values in other risk assessment documents, such as Health Assessment Documents (HADs), Health Effect Assessments (HEAs), Health and Environmental Effects Profiles (HEEPs), Health and Environmental Effects Documents (HEEDs), Drinking Water Criteria Documents (DWCDs), and Ambient Water Quality Criteria Documents (AWQCDs). Evaluations of potential carcinogenicity of chemicals in support of reportable quantity adjustments were published by EPA's Carcinogen Assessment Group (CAG) and may include cancer potency factor estimates. Health benchmarks derived by EPA for listing determinations or studies were also considered. Health benchmark values identified in these EPA documents are usually dated and are not recognized as Agency-wide consensus information or verified benchmarks.

ATSDR Minimal Risk Levels

The ATSDR MRLs are substance-specific health guidance levels for noncarcinogenic endpoints (ATSDR, 2003). An MRL is an estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse noncancer health effects over a specified duration of exposure. MRLs are based on noncancer health effects only and are not based on a consideration of cancer effects. MRLs are derived for acute, intermediate, and chronic exposure durations for oral and inhalation routes of exposure. Chronic inhalation and oral MRLs are similar to EPA's RfCs and RfDs, respectively (i.e., ATSDR uses the NOAEL/uncertainty factor approach); however, MRLs are intended to serve as screening levels and are exposure-duration-specific. Also, ATSDR uses EPA's inhalation dosimetry methodology (U.S. EPA, 1994) in the derivation of inhalation MRLs. When based on the same critical study, the MRLs have similar toxicity endpoints but may apply different uncertainty factors in contrast to EPA's RfDs and RfCs. MRLs are available on ATSDR's Web site and are presented in detail in individual ATSDR toxicological profiles. MRLs have undergone internal peer review and have been the subject of public comment.

CalEPA Cancer Potency Factors and Reference Exposure Levels

CalEPA has developed cancer potency factors for chemicals regulated under California's Hot Spots Air Toxics program (CalEPA, 1992, 1999). The cancer potency factors are analogous to EPA's oral and inhalation CSFs. CalEPA has also developed chronic inhalation reference exposure levels, analogous to U.S. EPA's RfCs, for 120 substances (CalEPA, 2000). CalEPA used EPA's inhalation dosimetry methodology (U.S. EPA, 1994) in the derivation of inhalation RELs. The cancer potency factors and inhalation RELs have undergone internal peer review by various California agencies and have been the subject of public comment.

7.2 Health Benchmark Values

Table 7-1 summarizes the chronic human health benchmarks used in this risk analysis. This table provides the Chemical Abstract Service Registry Number (CASRN), constituent name, cancer classification, RfD (in units of mg/kg-d), RfC (mg/m³), oral and inhalation CSF (mg/kg-d⁻¹), inhalation URF [(μg/m³)⁻¹], and reference for each benchmark. When available, maximum contaminant levels (MCLs) were also provided (U.S. EPA, 2002a). A key to the references cited and abbreviations used is provided at the end of the table.

Table 7-1. Chronic Health Benchmarks Used in the Dyes and Pigments Risk Assessment

Constituent	CASRN	RfD (mg/kg-d)	RfD Ref	RfC (mg/m ³)	RfC Ref	CSFo (per mg/kg-d)	CSFo Ref	URF (per µg/m ³)	URF Ref	CSFi (per mg/kg-d)	CSFi Ref	Notes
Aminoanthraquinone, 2-	117-79-3					3.3E-02	C99	9.4E-06	C99	3.3E-02	C99	
Aniline	62-53-3			1.0E-03	I	5.7E-03	I	1.6E-06	C99	5.7E-03	C99	
Anisidine, o-	90-04-0					1.4E-01	C92	4.0E-05	C92	1.4E-01	C92	
Azobenzene	103-33-3					1.1E-01	I	3.1E-05	I	1.1E-01	H	
Barium	7440-39-3	7.0E-02	I	5.0E-04	H							
Benzaldehyde	100-52-7	1.0E-01	I									
Benzidine	92-87-5	3.0E-03	I			2.3E+02	I	6.7E-02	I	2.3E+02	H	
Bis(dimethylamino) benzophenone, 4-4	90-94-8					8.6E-01	C99	2.5E-04	C99	8.6E-01	C99	
Chloroaniline, 4- (p-)	106-47-8	4.0E-03	I			5.4E-02	SF					
Copper	7440-50-8	*MCL only										MCL= 1.3 mg/L
Cresidine, p-	120-71-8					1.5E-01	C99	4.3E-05	C99	1.5E-01	C99	
Cresol, p- (4-methylphenol)	106-44-5	5.0E-03	H	6.0E-01	C00							RfC is for mixed cresols
Dichlorobenzene, 1,2- (o-)	95-50-1	9.0E-02	I	2.0E-01	H							
Dichlorobenzidine, 3,3'-	91-94-1					4.5E-01	I	3.4E-04	C99	1.2E+00	C99	
Dimethoxybenzidine, 3,3'-	119-90-4					1.4E-02	H					
Dimethylaniline, 2,4-	95-68-1					7.5E-01	H					
Dimethylbenzidine, 3,3'-	119-93-7					2.3E+00	SF					
Diphenylamine, N,N-	122-39-4	2.5E-02	I									

(continued)

Table 7-1. (continued)

Constituent	CASRN	RfD (mg/kg-d)	RfD Ref	RfC (mg/m ³)	RfC Ref	CSFo (per mg/kg-d)	CSFo Ref	URF (per µg/m ³)	URF Ref	CSFi (per mg/kg-d)	CSFi Ref	Notes
Formaldehyde	50-00-0	2.0E-01	I					1.3E-05	I	4.5E-02	H	
Lead and cmpds (inorganic)	7439-92-1	*MCL only										MCL = 0.015 mg/L
Methanol	67-56-1	5.0E-01	I	4.0E+00	C00							
Methyl-5-nitroaniline, 2-	99-55-8					3.3E-02	H					
Methylaniline, 2- (o-toluidine)	95-53-4					2.4E-01	H	6.9E-05	AC	2.4E-01	AC	
Methylene-bisbenzeneamine, 4,4'- (4,4'-methylenedianiline)	101-77-9	5.0E-02	I	2.0E-02	C00	1.6E+00	C99	4.6E-04	C99	1.6E+00	C99	surrogate RfD for Bisphenol A used as recommended by NCEA
Naphthalene	91-20-3	2.0E-02	I	3.0E-03	I							
Nitro-o-anisidine, 5-	99-59-2					4.6E-02	H	1.4E-05	C92	4.9E-02	C92	
N-N-Dimethylaniline	121-69-7	2.0E-03	I									
Phenol	108-95-2	3.0E-01	I									
Phenylenediamine, m-	108-45-2	6.0E-03	I									
Phenylenediamine, o-	95-54-5					4.7E-02	H					
Phenylenediamine, p-	106-50-3	1.9E-01	H									
Sodium nitrite	7632-00-0	1.0E-01	I									
Toluene-2,4-diamine (2,4-diaminotoluene)	95-80-7					3.2E+00	H	1.1E-03	C99	4.0E+00	C99	
Toluidine, p-	106-49-0					1.9E-01	H					
Zinc	7440-66-6	3.0E-01	I									

(continued)

Table 7-1. (continued)

Key:

CASRN	=	Chemical Abstract Service Registry Number	CSFo	=	oral cancer slope factor
RfD	=	reference dose	CSFi	=	inhalation cancer slope factor
RfC	=	reference concentration	URF	=	inhalation unit risk factor
MCL	=	maximum contaminant level			

- ^a Sources:
- A = ATSDR MRLs (ATSDR, 2003)
 - AC = developed for the Air Characteristic Study (U.S. EPA, 1999)
 - Cal00 = CalEPA chronic REL (CalEPA, 2000)
 - Cal99 = CalEPA cancer potency factor (CalEPA, 1999)
 - Cal92 = CalEPA cancer potency factor (CalEPA, 1992)
 - H = HEAST (U.S. EPA, 1997)
 - I = IRIS (U.S. EPA, 2003)
 - SF = Superfund Risk Issue Paper (U.S. EPA, 2002b, 2002c).

7.3 References

- ATSDR (Agency for Toxic Substances and Disease Registry). 2003. Minimal Risk Levels (MRLs) for Hazardous Substances. <http://www.atsdr.cdc.gov/mrls.html>
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U.S. EPA (Environmental Protection Agency). 2002c. *Provisional Toxicological Values for the Health Effects Assessment Summary Table (HEAST): 3,3'-Dimethylbenzidine (CASRN 119-93-7)*. National Center for Environmental Assessment, Superfund Technical Support Center, Cincinnati, OH. SRC TR-02-039/09-26-2002.

U.S. EPA (Environmental Protection Agency). 2003. Integrated Risk Information System (IRIS). National Center for Environmental Assessment, Office of Research and Development, Washington, DC. Available online at <http://www.epa.gov/iris/>

8.0 Human Health Risk Estimation

The final step of the risk assessment process is to estimate the risk posed to receptors. In this step, the preceding components of the risk assessment—estimates of toxicity (the health benchmarks) and exposure assessments—are summarized and integrated into quantitative expressions of risk. Estimates of dose and toxicity are used to calculate individual excess lifetime carcinogenic risk estimates and noncancer HQs for constituents. This section describes the risk calculations completed for this analysis that were based on unit waste concentrations (e.g., 1 mg/kg).

The goal of this risk assessment was to generate risk-based allowable mass loadings of a constituent that can be present in waste and remain below a specified target risk level. To accomplish this, the risk of managing wastes in each of the WMUs (landfills, surface impoundments, and tanks) first had to be predicted. Thus, from the probabilistic analysis, a distribution of risk estimates was generated based on a single unit waste concentration.

The target risk level for this assessment was either

- An excess individual lifetime cancer risk of 1 chance in 100,000 of developing cancer ($1E-5$) for constituents that can produce cancer health effects
- or
- A ratio of projected intake levels to safe intake levels, an HQ, of 1 for constituents that can produce noncancer health effects.

Not only can exposure to a constituent create both cancer and noncancer health impacts, but the type and magnitude of the exposure will differ depending on whether the constituent was ingested or inhaled. As such, the cancer and noncancer health impacts were calculated for both ingestion and inhalation of the constituent. Because there is a different risk resulting from the type of health impact (cancer vs. noncancer) and route of exposure (ingestion vs. inhalation), different risk endpoints were generated for each constituent in each WMU. Table 8-1 lists the risk endpoints.

The risks resulting from exposures via the air pathway and groundwater pathway were evaluated separately. Estimated exposures from air pathways occur during the operating or postclosure life of the unit, whereas risks via the groundwater pathways are, for the most part, not projected to occur within the same timeframe. Therefore, risks from the groundwater pathways of ingestion of drinking water and inhalation of shower emissions are added, but risk from inhalation of ambient air is considered separately. As a result, the risk endpoints were estimated twice, once for aboveground exposure and once for groundwater exposure. Sections 8.1 through 8.4 provide further details on how each risk endpoint was determined.

Table 8-1. Risk Endpoints Used for Risk Categories

Risk Category	Risk Endpoints	Definition
Carcinogens	Lifetime excess cancer risk - inhalation	Lifetime excess cancer risk resulting from inhalation exposure to a single chemical
	Lifetime excess cancer risk - ingestion	Lifetime excess cancer risk resulting from ingestion exposure to a single chemical
	Total lifetime excess cancer risk	Lifetime excess cancer risk resulting from multiple pathway exposures to a single chemical
Noncarcinogens	Ingestion HQ by pathway	Ingestion pathway noncancer risk characterization for a single chemical for a single ingestion pathway component (e.g., soil ingestion)
	Ingestion HQ	Ingestion pathway noncancer risk characterization from exposure to all ingestion pathway components for a single chemical
	Inhalation HQ	Inhalation pathway noncancer risk characterization for a single chemical

8.1 Lifetime Excess Cancer Risk

Cancer risk was characterized using lifetime excess cancer risk estimates to represent the excess probability of developing cancer over a lifetime as a result of exposure to the COC. Lifetime excess cancer risk estimates use the LADD as the exposure metric. Lifetime excess cancer risk estimates are the product of the LADD for a specific receptor and the corresponding CSF, as shown in Equation 8-1. Lifetime excess cancer risk estimates were calculated separately for inhalation and ingestion exposures because they are based on separate routes of exposure and use different CSFs.

$$\text{Lifetime excess cancer risk} = \text{LADD} \times \text{CSF} \quad (8-1)$$

where

LADD = lifetime average daily dose (mg/kg BW/d)
 CSF = cancer slope factor (mg/kg BW/d)⁻¹.

8.2 Total Lifetime Excess Cancer Risk

Constituent-specific individual lifetime excess cancer risks were generated for each receptor for inhalation and ingestion pathway exposures, and then these pathway-specific lifetime excess cancer risks were summed to generate a total lifetime excess cancer risk for each receptor. Thus, risks due to inhalation of shower emissions were added to risks due to ingestion

of drinking water. As noted above, risks due to inhalation of ambient air were considered separately because these exposures typically do not occur in the same timeframe.

8.3 Ingestion Hazard Quotient by Pathway

Noncancer risk is characterized through the use of HQs, which use the ADD as the exposure metric. HQs are calculated by dividing an ADD for ingestion pathways by the corresponding RfD. An HQ establishes whether a particular individual has experienced exposure that places him or her either above or below a threshold of concern for a specific health effect. Therefore, unlike cancer risk estimates, HQs are not probability statements. Rather, the RfD represents a “no-effects” level that is presumed to be without appreciable risk from chronic

$$HQ_{\text{ingest}} = \frac{ADD_{\text{ingest}}}{RfD} \quad (8-2)$$

exposures over a lifetime. The RfD may be derived from human or animal studies and may include uncertainty factors to account for deficiencies in the available studies. Equation 8-2 shows the calculation for the ingestion HQ.

where

ADD = average daily dose for ingestion pathway (mg/kg-d)
RfD = reference dose (mg/kg-d).

8.4 Inhalation Hazard Quotient

The inhalation HQ is similar to the ingestion HQ in that it represents a ratio of an exposure to a reference value. However, unlike the ingestion HQ, which uses the ADD as the exposure metric, the inhalation HQ uses an air concentration as the exposure metric. This concentration is compared to a reference concentration. As with the RfD, the RfC represents a

$$HQ_{\text{inhal}} = \frac{C_{\text{air}}}{RfC} \quad (8-3)$$

“no-effects” level that is presumed to be without appreciable risk of adverse effects from chronic exposures over a lifetime. The RfC may be derived from human or animal studies and may include uncertainty factors to account for deficiencies in the available studies. Inhalation HQ is calculated as follows:

where

C_{air} = ambient air concentration (mg/m³)
RfC = reference concentration (mg/m³).

9.0 Allowable Mass Loading Calculations

The objective of the dyes and pigments risk assessment was to calculate chemical-specific mass loading limits in kilograms per year (kg/yr) that reflect an annual quantity of a constituent that can be placed in a landfill, surface impoundment, or tank and be protective of human health and the environment. These mass loading limits are defined as mass loadings that result in cancer risks no greater than 1 in 100,000 (1E-5) and noncancer HQs no greater than 1 at the 90th percentile of the distribution of risks. Thus, when the mass loading limit for a constituent is modeled for a distribution of WMUs, the resulting risk distribution will show that the 90th percentile cancer risk is 1E-05 or the 90th percentile HQ is 1.

The annual mass loading was calculated for each organic constituent as shown in Equation 9-1.

$$\text{ConstitLoad} = \text{Conc}_{\text{constit}} \times \text{Frac}_{\text{dp}} \times \text{WMU}_{\text{load}} \times \text{BD} \times 0.000001 \quad (9-1)$$

where

ConstitLoad	=	loading for a constituent (kg/yr)
Conc _{constit}	=	concentration of constituent in dye and pigment waste (mg/kg)
Frac _{dp}	=	fraction of the waste in the WMU that is dye and pigment waste (unitless)
WMU _{load}	=	annual load (all waste) to WMU (m ³ /yr)
BD	=	bulk density of the waste (kg/m ³)
0.000001	=	units conversion factor (kg/mg).

The methodology to determine the loading for a constituent is described separately for organic constituents and metals because the entire system is linear for organic constituents, but the groundwater model (EPACMTP) is not linear for metals. The methodology described in this section applies to landfills, but the same method applies to surface impoundments and tanks, as well. For surface impoundments and tanks, annual total influent flow is analogous to the annual landfill load.

9.1 Organic Constituents

For organic constituents, the general approach follows an 8-step procedure:

1. Set the initial constituent mass loadings.

The first step is to determine an initial mass loading. This initial mass loading is defined as the mass of constituent contained in the smallest landfill when that landfill is assumed to be filled with dye and pigment waste containing a unit concentration (1 mg/kg) of that constituent. Therefore, the smallest landfill in the distribution was identified, and the fraction of waste that is dye and pigment waste was set equal to 1. A landfill is assumed to be filled to capacity over a period of 30 years; therefore, the total WMU volume was divided by 30 years to yield an annual waste volume. Then, the initial constituent mass loading was calculated using Equation 9-2:

$$\text{ConstitLoad}_{\text{initial}} = \text{WMUload}_{\text{min}} \times \text{Conc}_{\text{constit}} \times \text{BD} \times 0.000001 \quad (9-2)$$

where

$\text{ConstitLoad}_{\text{initial}}$	=	initial loading for a constituent (kg/yr)
$\text{Conc}_{\text{constit}}$	=	concentration of constituent in dye and pigment waste (mg/kg)
$\text{WMUload}_{\text{min}}$	=	annual load (all waste) necessary to fill smallest WMU (m ³ /yr)
BD	=	bulk density of waste (kg/m ³)
0.000001	=	units conversion factor (kg/mg).

2. Calculate the fraction of waste at each WMU that is dye and pigment waste.

For the smallest landfill, this fraction will be 1. For all others, it will be less than 1. The fraction of waste that is dye and pigment waste was calculated for all other landfills in the Monte Carlo simulation using Equation 9-3.

$$\text{Frac}_{\text{dp}} = \frac{\text{WMUload}_{\text{min}}}{\text{WMUload}} \quad (9-3)$$

where

Frac_{dp}	=	fraction of waste that is dye and pigment waste
$\text{WMUload}_{\text{min}}$	=	annual load (all wastes) necessary to fill the smallest WMU (kg/yr)
WMUload	=	annual capacity (kg/yr) of a particular WMU in the Monte Carlo simulation.

By determining the initial concentration of the constituents and the fraction of waste that is dye and pigment waste, a constant mass load to be managed in each landfill could be set in the Monte Carlo simulation.

3. Run the landfill model using the initial load and waste fractions from steps 1 and 2, respectively.

The landfill source model was run for 10,000 iterations to address variability in site location and WMU characteristics. Each iteration produced a 200-year time series for annual leachate flux and volatile air emissions.

4. Calculate exposure point concentrations for groundwater and ambient air pathways.

- a. The EPACMTP model was run to generate average groundwater well concentrations corresponding to the exposure duration of each receptor considered. The 10,000 iterations of the 200-year leachate flux time series from the source model were first used as input to the groundwater model in order to generate a time series of groundwater well concentrations. For each iteration of the model, well concentrations were averaged around the peak groundwater well concentration in the time series using exposure durations for each receptor (i.e., adult and child) from a distribution of 10,000 exposure durations.
- b. Ground-level ambient air concentrations at receptor locations were estimated using the volatile emissions from the landfill. The 10,000 iterations of the 200-year air emissions time series were paired with UACs from the IWAIR UAC database. The IWAIR data provide maximum UACs for volatile emissions. These data were developed for IWAIR to provide a national distribution of UACs at a specified distance from the source. A distribution of 10,000 distances from the WMU was generated using the distribution of distances. The IWAIR UAC ($\mu\text{g}/\text{m}^3$ per $\text{g}/\text{m}^2\text{-s}$) multiplied by the volatile emission flux ($\text{g}/\text{m}^2\text{-s}$) yields the maximum air concentration ($\mu\text{g}/\text{m}^3$) for a given iteration.

5. Run the risk model using ambient air and groundwater exposure point concentrations to generate cumulative distributions of risks and HQs for each chemical, WMU, and receptor (e.g., for aniline, in unlined landfills, for the child receptor).

The 10,000 groundwater well concentrations resulting from EPACMTP modeling and the 10,000 iterations of air concentration data at receptor locations were used as inputs to the risk model to estimate risk or hazard. Three pathways were included in the exposure and risk modeling: inhalation of ambient air, ingestion of groundwater, and inhalation of constituents in groundwater during showering. The exposure and risk model generated a distribution of risk estimates for all three exposure pathways for both child and adult receptors (only adults were assessed for showering) for all applicable endpoints (i.e., cancer, noncancer). Total risks from the groundwater pathway were calculated by summing the risks due to inhalation of shower emissions and risks due to ingestion of drinking water. Risks due to inhalation of ambient air were considered separately because these exposures typically do not occur in the same timeframe.

- 6. Select the pathway/receptor combination showing the greatest risk at the 90th percentile for the constituent/WMU/liner combination (e.g., ingestion of drinking water/child receptor). This is called the limiting pathway.**

The 90th percentile risk or hazard was selected for each receptor and pathway combination from each of the distributions of risk and then compared with 90th percentile values to determine the pathway and receptor with the highest risk at the 90th percentile. These are the risk-limiting pathway, receptor, and endpoint.

- 7. Calculate the ratio of the target risk and HQ (i.e., risk of 1E-05 or HQ of 1) to the 90th percentile modeled risk or HQ for the limiting pathway (pathway with the highest risk).**

The highest risk or hazard determined in step 6 was compared with the target risk (risk of 1E-05 or HQ of 1) to determine the ratio of target risk to calculated risk (see Equation 9-4). This ratio is called the scaling factor and is used to calculate the mass loading limits.

$$\text{Scaling Factor} = \text{Risk}_{\text{target}} / \text{Risk}_{\text{initial}} \quad (9-4)$$

where

Scaling Factor	=	scaling factor to determine mass loading limit (unitless)
Risk _{target}	=	target risk = 1E-05 or HQ = 1 (unitless)
Risk _{initial}	=	90 th percentile risk modeled based on initial mass loading (unitless).

- 8. Calculate mass loading limits.**

Equation 9-5 was used to calculate the target mass loading for each chemical/WMU/receptor:

$$\text{ConstitLoad}_{\text{target}} = \text{Scaling Factor}_{\text{target}} \times \text{ConstitLoad}_{\text{initial}} \quad (9-5)$$

where

ConstitLoad _{target}	=	constituent loading that results in target HQ (kg/yr)
Scaling Factor	=	scaling factor calculated in step 7 (unitless)
ConstitLoad _{initial}	=	initial constituent loading (kg/yr).

The result is the mass loading limit for each chemical/WMU/liner type scenario.

9.2 Metals

The groundwater model (EPACMTP) includes the use of nonlinear K_d isotherms for metal constituents. Therefore, risks and HQs for metals are not linearly related to mass loadings. The four metals in the analysis (barium, copper, lead, and zinc) all have noncancer endpoints. In

addition, metals are nonvolatile, and thus no inhalation pathways (either ambient air or shower exposure) are modeled. Consequently, a different procedure was used to calculate the mass loading limit for metals. This procedure consisted of a 5-step approach, as follows:

1. Run source model.

The first step in the technical approach to develop mass loading limits for metals is to execute the source model using the same initial mass load and fraction dye and pigment wastes used for organic constituents. The resulting 10,000 iterations of 200-year of leachate flux time series were used to execute EPACMTP.

2. Assume an initial groundwater well concentration, run the exposure and risk model, and select the 90th percentile value.

Parallel to step 1, the exposure and risk model was run using an initial well water concentration of 1 µg/L to obtain a distribution of HQs for the initial well concentration. Then, the 90th percentile HQ was selected from the resulting distribution of HQs.

3. Scale the initial well water concentration (from step 2 above) by the ratio of the target HQ (i.e., 1) to the modeled 90th percentile HQ to obtain a concentration that corresponds to the target 90th percentile HQ as follows:

$$WC_{90} = WC_{\text{initial}} \times (HQ_{\text{target}} / HQ_{90}) \quad (9-6)$$

where

WC_{90}	=	well concentration that will result in a risk distribution with a 90 th percentile of 1 (µg/L)
WC_{initial}	=	initial well concentration (µg/L)
HQ_{target}	=	target HQ (unitless)
HQ_{90}	=	90 th percentile HQ for initial well concentration (unitless).

4. Execute EPACMTP model in an iterative mode to determine the leachate flux scaling factor that will result in a well water concentration that corresponds to the target 90th percentile HQ.

- a. The EPACMTP was run using the leachate concentrations generated by the source model to determine a distribution of 10,000 groundwater concentrations.
- b. The 90th percentile groundwater concentration was then compared to WC_{90} , and the ratio of the 90th percentile groundwater concentration to WC_{90} was calculated. This ratio was used to adjust the entire leachate flux distribution from step 1, and then the EPACMTP model was run again using this adjusted leachate flux distribution.
- c. This process was repeated until a scaling factor that resulted in the 90th percentile groundwater concentration equaling WC_{90} was determined. The final scaling

factor determined by this iterative EPACMTP modeling process is the ratio of calculated leachate flux to the initial leachate flux.

5. Multiply initial mass loading to the landfill by the scaling factor to determine the mass loading limit.

The mass loading limit was calculated from the initial mass loading and the scaling factor as follows:

$$\text{ConstitLoad}_{\text{target}} = \text{ConstitLoad}_{\text{initial}} \times \text{Scaling Factor} \quad (9-7)$$

where

$\text{ConstitLoad}_{\text{target}}$ = constituent loading that results in target HQ (kg/yr)
 $\text{ConstitLoad}_{\text{initial}}$ = initial constituent loading (kg/yr)
Scaling Factor = scaling factor calculated in step 4 (unitless).

10.0 Variability and Uncertainty

This section discusses the methods that were used in the dyes and pigments listing risk assessment to account for variability and uncertainty. Variability and uncertainty are discussed separately because they are fundamentally different. Variability represents true heterogeneity in characteristics, such as body weight differences within a population or differences in contaminant levels in the environment. It accounts for the distribution of risk within the exposed population. Uncertainty, on the other hand, represents lack of knowledge about factors, such as adverse effects from contaminant exposure, which may be reduced with additional research to improve data or models.

Variability arises from true heterogeneity in characteristics, such as body weight differences within a population or differences in contaminant levels in the environment.

Uncertainty represents lack of knowledge about factors, such as the nature of adverse effects from exposure to constituents, which may be reduced with additional research.

This discussion describes the treatment of variability and uncertainty in reference to some parameters used to describe human exposures and risk. Treatment of variability using a Monte Carlo simulation forms the basis for the human health risk distributions, which in turn are the basis for calculating protective waste and leachate concentrations. Previous sections of this technical background document describe how distributions were generated and point values estimated for input parameters. They also describe how these values were used in the models and calculations to produce national level distributions of waste and leachate concentrations that are protective of human health. Uncertainty necessitated the use of assumptions, default values, and imputation techniques in this study. This discussion focuses on how this treatment of variability and uncertainty affects the results.

10.1 Variability

Variability is often used interchangeably with the term uncertainty, but the two are not the same. Variability is tied to variations in physical, chemical, and biological processes and cannot be reduced with additional research or information. Although variability may be known with great certainty (e.g., age distribution of a population may be known and represented by the mean age and its standard deviation), it cannot be eliminated and needs to be treated explicitly in the analysis. Spatial and temporal variability in parameter values used to model exposure and risk account for the distribution of risk in the exposed population.

For example, the meteorological parameters used in dispersion modeling, such as windspeed and wind direction, are measured hourly by the National Weather Service at many

locations throughout the United States, and statistics about these parameters are well documented. Although the distributions of these parameters may be well known, their actual values vary spatially and temporally and cannot be predicted exactly. Thus, the concentration calculated by a dispersion model for a particular receptor for a particular time period will provide information on average conditions that may over- or underpredict actual concentrations. Much of the temporal variation is accounted for by using models such as the ISCST3 that calculate concentrations hourly and sum these hourly values to provide annual concentration estimates. Additionally, using meteorological data from multiple monitoring stations located throughout the United States can account for some but not all spatial variability.

This analysis was designed to specifically address as much of the variability as possible, either directly in the Monte Carlo analysis or through disaggregation of the data into discrete elements of the analysis. For example, variability in WMU characteristics was accounted for by using large databases of individual WMU characteristics that represent the range of possible WMU characteristics. Spatial variability in environmental setting was accounted for by using different locations around the contiguous 48 states. Because dye and pigment facilities, and therefore the disposal of wastes generated during manufacturing, occur nationally, this analysis used regional databases to characterize environmental conditions that influence the fate and transport of constituents in the environment.

The risk assessment components discussed include

- Source characterization and emissions modeling
- Fate and transport modeling
- Exposure modeling.

10.1.1 Source Characterization and Emissions Modeling

The specific WMUs in which dye and pigment wastes are disposed were not known; however, EPA determined that wastes could be disposed of in landfills, surface impoundments, or tanks. For this analysis, national databases containing information on various WMUs and their design and operating characteristics were used to characterize the variability in WMUs. Landfills were characterized using EPA's Solid Waste (Municipal) Landfill Survey (U.S. EPA, 1988) for area and depth. Other parameters needed to characterize landfills were obtained from the *Industrial Subtitle D Survey* (Schroeder et al., 1987). The *Industrial Subtitle D Survey* (Schroeder et al., 1987) was also used to characterize surface impoundments, and TSDf survey data (U.S. EPA, 1987) were used to characterize treatment tank characteristics. Using the information contained in these databases, three distributions of WMUs were developed, one for each WMU type. These distributions were used in the probabilistic analysis to capture the national variation in WMU physical and operating characteristics.

Source partition modeling was performed for different locations, which allowed variation in location-dependent parameters (e.g., soil, temperature, precipitation) to be considered explicitly in the modeling. Variation in these parameters influenced variation in predicted air emissions, leachate, and infiltration rates. Statistical distributions were used to vary the values of many model input parameters used to characterize WMU and waste characteristics. Meteorological data sets containing UAC values were combined with these WMU physical

characteristics data (e.g., surface area) and emissions data to estimate air concentrations for landfills, surface impoundments, and tanks.

In the Monte Carlo analysis, the WMU characteristics and locations were randomly selected from national databases to produce the 10,000 iterations of the source model calculations. The source model calculations generated the distribution of environmental releases used in the fate and transport modeling.

10.1.2 Fate and Transport Modeling

The parameter values required to model contaminant fate and transport were obtained from regional databases. The following sections discuss the treatment of regional variation in location-dependent parameters used in fate and transport modeling.

10.1.3 Receptor Location

The location of receptors was an important source of variability addressed in the exposure modeling. Previous EPA studies provided data on distances between WMUs and nearest human receptors and on distances between landfills and nearest residential wells. Because EPA is interested in protecting people residing near WMUs, these data were used to develop distributions for distance to receptor to capture variation in resident location. Individuals may potentially be located in any direction and at various distances from a facility, and this analysis explicitly incorporated this consideration. For the air pathways, a receptor grid was established to locate individuals at varying distances between 50 and 550 m from the edge of the WMU. The Monte Carlo analysis used a normal distribution to assign probability to various distances from the WMU, giving greater weight to the central tendency distance of 300 m, and a uniform distribution to assign direction, giving equal probability to a receptor being located in any direction. For the groundwater pathway, downgradient distance and the location within the lateral extent of the groundwater plume were varied.

10.1.4 Air Dispersion Modeling

To capture geographic variation, dispersion modeling was conducted using meteorological data sets from 60 different meteorological stations around the contiguous 48 states. This provided regional representation of the variability in meteorological data. For landfills and surface impoundments, these data sets were combined with surface areas representing the distribution of WMU size to provide different sets of UACs to use with emissions data to estimate air concentrations. For tanks, these data sets were combined with surface area-height combinations representing the distribution of WMU dimensions to provide different sets of UACs to use with emissions data to estimate air concentrations.

10.1.5 Groundwater Modeling

To capture regional differences in aquifer types for use in the groundwater modeling, aquifers typical of the meteorological station locations were characterized. For each location, aquifer types typical of the region were identified. If more than one aquifer type was associated with a given location, equal weight was assigned to each aquifer type for use in groundwater

modeling. This approach captured the national distribution of aquifer types and ensured that all aquifer types were included in the modeling.

Within each aquifer type, aquifer characteristics (e.g., aquifer thickness or vadose zone depth) are variable. To account for this variability, vadose parameters were varied within each aquifer type using data from EPACMTP. Correlated aquifer parameters were varied together for each aquifer type to preserve the correlation of those parameters.

10.1.6 Exposure Modeling

Individual physical characteristics, activities, and behavior are quite different. As such, the exposure factors that influence the exposure of an individual, including inhalation rate, ingestion rate, body weight, and exposure duration, are quite variable. To include this variability explicitly in the analysis, statistical distributions for these variables were used for each receptor in the analysis: adult and child residents. For adults, a single exposure factor distribution was used for males and females. For child exposures, one age group (ages 1 to 5) was considered, representing age at the start of exposure, because for most health effects, this age group is most sensitive. Exposure parameter data were used from the EFH (U.S. EPA, 1997a,b,c) to establish statistical distributions of values for each exposure parameter for each receptor.

10.1.7 Summary of Variability Considerations

In summary, a distribution of protective waste and leachate concentrations was developed that specifically considers the variability in

- WMU and waste characteristics
- Location of receptors
- Regional-specific environmental conditions
- Exposure factors for each receptor.

Taken together, these provide national distributions of a risk-specific waste and leachate concentration across all facilities of a specified type.

10.2 Uncertainty

Uncertainty is a description of the imperfection in knowledge of the true value of a particular parameter. In contrast to variability, uncertainty is reducible by additional information gathering or analysis activities (i.e., better data, better models). The major areas of uncertainty in risk assessments are typically classified as scenario uncertainty, model uncertainty, and parameter uncertainty. Scenario uncertainty refers to missing or incomplete information needed to fully define exposure and dose. Model uncertainty is a measure of how well the model simulates reality. Parameter uncertainty is the lack of knowledge regarding the true value of a parameter used in the analysis.

Although some aspects of uncertainty are directly addressed in this analysis, much of the uncertainty associated with this analysis could only be addressed qualitatively. This section presents significant sources of uncertainty. If the analysis directly addressed uncertainty, the

approach used is described. If the analysis did not directly address uncertainty, a qualitative discussion of its importance is provided.

10.2.1 Scenario Uncertainty

Sources of scenario uncertainty include the assumptions and modeling decisions that are made to represent an exposure scenario. The lack of information or resources to define and model actual exposure conditions introduced uncertainty into this analysis.

Professional judgment was used, often coupled with an evaluation of the results of sensitivity analyses, to decide which parameters to include in describing exposure conditions and behaviors. Scenario uncertainties that are important to understand in interpreting the results of this study are discussed in the following subsections.

10.2.1.1 Waste Characteristics. Very few data were available on the physical and chemical characteristics of dye and pigment waste. To address this lack, assumptions were made on the waste characteristics based on general knowledge of generic industrial wastes. In this analysis, except for constituent mass, which was calculated, it was assumed that the waste in the WMU mixed with other wastes. Therefore, general waste characteristics were used, including default assumptions for the waste parameters (e.g., bulk density, moisture, pH).

10.2.1.2 Receptor Populations Evaluated. Risk estimates presented in this document address hypothetical chronic exposures for these receptors and are designed to provide a realistic range of potential scenarios. Not all potential scenarios were evaluated; for example, infants (0- to 1-year-olds) were not evaluated.

10.2.1.3 Exposure Uncertainty. Exposure modeling relies heavily on default assumptions concerning population activity patterns, mobility, dietary habits, body weights, and other factors. As described earlier in the variability section, the Monte Carlo analysis for the adult and child exposure scenario addressed the possible variability in the exposure modeling by using distributions of values for exposure factors. There are some uncertainties, however, in the data that are used. Although it is possible to study various populations to determine various exposure parameters (e.g., age-specific soil ingestion rates or intake rates for food) or to assess past exposures (epidemiological studies) or current exposures, risk assessment is about prediction. Therefore, long-term exposure monitoring in this context is infeasible. The EFH (U.S. EPA, 1997a,b,c) provides the current state-of-the-science concerning exposure modeling and assumptions and is used throughout this document. To the extent that actual exposure scenarios vary from the assumptions in this risk assessment, risks could be underestimated or overestimated. However, although there could be individuals living near a WMU who have higher exposures than those predicted, it is more likely that actual exposures for most of these individuals would fall within the predicted range and, moreover, would be similar to what was modeled.

10.2.1.4 Natural Background Exposures. In certain cases, a risk assessment is performed on wastes that contain contaminants that also are present in the environment as a result of both natural processes and anthropogenic activities. Under these circumstances, receptors potentially receive a “background” exposure that may be greater than the exposure

resulting from release of contaminants from the waste. For national analyses like this assessment, the inclusion of background concentrations as part of the analysis is not feasible because of the variability of background concentrations nationwide and the lack of data on national background concentrations for each constituent. Not including the exposure an individual may already have to a COC (i.e., exposure to background concentrations) does not change the “marginal” increase in risk a person may have as a result of possible exposures to constituents in dye and pigment waste.

10.2.2 Model Uncertainty

Model uncertainty is associated with all models used in all phases of a risk assessment because models and their mathematical expressions are simplifications of reality that are used to approximate real-world conditions and processes and their relationships. Computer models are simplifications of reality, requiring exclusion of some variables that influence predictions but that cannot be included in models either because of increased complexity or because of a lack of data on a particular parameter. Models do not include all parameters or equations necessary to express reality because of the inherent complexity of the natural environment and the lack of sufficient data to describe the natural environment. Because this is a probabilistic assessment that predicts what may occur with the management of certain dye and pigment wastes under assumed scenarios, it is not possible to compare the results of these models to any specific situation that may exist. The risk assessor needs to consider the importance of excluded variables on a case-by-case basis because a given variable may be important in some instances and not in others. A similar problem can occur when a model that is applicable under average conditions is used for conditions that differ from the average. In addition, in some instances, choosing the correct model form is difficult when conflicting theories seem to explain a phenomenon equally well. In other instances, EPA-approved model forms for addressing certain phenomena, such as facilitated transport, are not available. Models used in this risk assessment were selected based on science, policy, and professional judgment. These models were selected because they provide the information needed for this analysis and because they are generally considered to be state-of-the-science. Even though the models used in the risk analyses are used widely and have been accepted for numerous applications, they each retain significant sources of uncertainty. Evaluated as a whole, the sources of model uncertainty in this analysis could result in either an overestimation or underestimation of risk. Specific areas of modeling uncertainty in this analysis are as follows:

- There were multiple constituents identified as materials used in dye and pigment manufacturing that were not modeled in this risk assessment because of a lack of information on how they behave when introduced to the environment. The fate and transport modeling was limited to those constituents for which (1) the physical-chemical parameters necessary to run the models were available and (2) adequate information on toxicity was available to estimate potential health impacts from exposure.
- Exposure modeling relies heavily on default assumptions concerning population activity patterns, mobility, dietary habits, body weights, and other factors. There are some uncertainties associated with some of the data used for these parameters. Although it is possible to study various populations to determine various exposure

parameters or to assess past exposures (epidemiological studies) or current exposures, risk assessment is about prediction. Therefore, long-term exposure monitoring in this context is infeasible. The EFH, which provides the current state-of-the-science concerning exposure modeling and assumptions, was used in this risk assessment. To the extent that actual exposure factors vary from the assumptions in this risk assessment, risks could be underestimated or overestimated.

- In modeling the fate and transport of chemicals in groundwater, complex hydrogeology, such as karst or highly fractured aquifers, was not assessed. Some fraction of the groundwater settings in this analysis have fractured flow. In general, fractured flow in groundwater can channel the contaminant plume, thus allowing it to move faster and in a more concentrated state than in a nonfractured flow environment. As a result, the modeling may under- or overestimate the concentrations in the groundwater.
- There is uncertainty in predicting the movement of contaminants over long periods of time. The risk to receptors for the groundwater pathway was evaluated over a time period of 10,000 years. There are significant uncertainties concerning how exposure and environmental assumptions will change over time, and the modeling methodology does not change these assumptions over this 10,000-year period.

10.2.2.1 Air Dispersion Modeling. The ISCST3 model was used to calculate the dispersion vapor emissions from a WMU. This model has many capabilities needed for this assessment, such as the ability to model area sources. For dispersion modeling of this type, ISCST3 is considered to be a fairly accurate model with error within about a factor of 2. It does not include photochemical reactions or degradation of a chemical in the air, which results in additional model uncertainty for some chemicals.

10.2.2.2 Groundwater Modeling. In the groundwater model, EPACMTP, it is assumed that the soil and aquifer are uniform porous media. EPACMTP does not model preferential pathways, such as fractures, macropores, or facilitated transport, which may affect migration of strongly sorbing constituents such as metals. EPACMTP also does not model colloidal transport or the geochemical interactions between different contaminants in the leachate. Any of these factors could result in underpredicting contaminant concentrations at the receptor well. Conversely, the EPACMTP modeling incorporates the following assumptions: (1) transverse dispersion is negligible in the unsaturated zone, potentially resulting in an overestimation of risks; (2) receptors use the uppermost aquifer, rather than a deeper aquifer, as a domestic source of drinking water, which overestimates risks where the uppermost aquifer is not used; and (3) hydrogeologic conditions that influence contaminant fate and transport are uniform spatially as well as temporally (that is, in the time period over which the model is executed, 10,000 years), potentially resulting in an underestimation or overestimation of receptor well concentrations.

10.2.2.3 Assumption of Additivity of Chemicals in Characterizing Risk. Both cancer and noncancer risks were evaluated on a chemical-specific basis within the analysis. Additive effects from multiple-chemical exposures were not calculated. Chemical mixtures can display

both synergistic and antagonist behavior with regard to risk. In general, however, the overall risk of a mixture is very likely to be greater than that of exposure to a single chemical. Therefore, not adding risks across the chemicals is an area of uncertainty that leads to an underestimate of total risk. The additive effects from multiple-chemical exposure were not calculated because the information available from the Toxics Release Inventory on the mass loading and comanagement of particular COCs in dye and pigment manufacturing wastes indicated that such codisposal in landfills was not a significant occurrence.

10.2.2.4 Human Health Benchmarks. Sources of uncertainty in toxicological benchmarks include one or more of the following: extrapolation from laboratory animal data to humans, variability of response within the human population, extrapolation of responses at high experimental doses under controlled conditions to low doses under highly variable environmental conditions, and adequacy of the database (number of studies available, toxic endpoints evaluated, exposure routes evaluated, sample sizes, length of study, etc.). Toxicological benchmarks are designed to be conservative (that is, to potentially overestimate risk) because of the uncertainties and challenges associated with condensing toxicity data into a single quantitative expression.

Cancer Slope Factors. CSFs were derived as the 95 percent lower confidence limit of the slope of the dose-response curve using a linear, no-threshold dose-response model. The CSF is, therefore, an upper-bound estimate of the cancer risk per unit dose and, for this reason, may overstate the magnitude of the risk. In addition, the use of CSFs in projecting excess individual cancer risk introduces uncertainty stemming from a number of factors, including

- Limited understanding of cancer biology
- Variability in the response of animal models
- Differential response in animal models versus humans
- Difference between animal dosing protocols and human exposure patterns.

A key step in CSF development is high- to low-dose extrapolation. Depending on the model used to fit the data, extrapolations to the low-dose range can vary by several orders of magnitude, reflecting the potential uncertainty associated with the CSF.

Reference Doses and Reference Concentrations. Uncertainty in the toxicological and epidemiological data from which RfDs and RfCs are derived is accounted for by applying uncertainty factors. An RfD (or RfC) is “an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime” (U.S. EPA, 2003). RfDs and RfCs are based on the NOAEL or lowest observed adverse effects level (LOAEL) for the most sensitive effect in the most sensitive or most relevant species. A series of standard uncertainty factors are applied to the NOAEL or LOAEL to derive the RfD or RfC. The following uncertainty factors account for areas of scientific uncertainty:

- Intraspecies variation—accounts for variation in sensitivity among humans (including sensitive individuals such as children, the elderly, or asthmatics)
- Interspecies variation—accounts for extrapolating from animals to humans

- LOAEL to NOAEL extrapolation
- Subchronic to chronic—accounts for extrapolating from a subchronic NOAEL or LOAEL to a chronic NOAEL or LOAEL
- Incomplete database—accounts for the lack of data for critical endpoints (e.g., reproductive and developmental).

Uncertainty factors of 1, 3, or 10 are used. The default value is 10; however, an uncertainty factor of 3 may be used, for example, if appropriate pharmacokinetic data (or models) are available. In addition, a modifying factor may be applied to account for additional uncertainties in accordance with professional judgment. The default value for the modifying factor is 1. All uncertainty factors (UFs) and the modifying factor (MF) are multiplied together to derive the total uncertainty factor (U.S. EPA, 1994). Therefore, the RfD (or RfC) is derived by using the following formula:

$$\text{RfD} = \text{NOAEL}/(\text{UF} \times \text{MF}). \quad (10-1)$$

The effect of applying uncertainty and modifying factors is to lower the estimate of the RfD and increase the HQ for a given exposure.

Human Health Benchmarks and Children. Significant uncertainties exist regarding the estimation of lifetime cancer risks in children. The risk of developing cancer was calculated from the estimated LADD and the slope of the dose-response curve. A CSF is derived from either human or animal data and is taken as the upper bound on the slope of the dose-response curve in the low-dose region, generally assumed to be linear, expressed as a lifetime excess cancer risk per unit exposure. Individuals exposed to carcinogens in the first few years of life may be at increased risk of developing cancer.

The noncancer toxicological effects in children are also an area of uncertainty. HQs for children are based on comparing childhood exposure, for which age-specific data (e.g., food consumption rates) are available, with adult toxicity measures (e.g., RfDs), for which adequate age-specific dose-response data are often lacking. This mismatch could result in great uncertainty in the estimation of HQs for children. This could sometimes result in an overestimation of children's risk and sometimes in an underestimation. This issue is still under investigation in the scientific community, and no consensus has been reached.

10.2.3 Parameter Uncertainty

Parameter uncertainty occurs when (1) there is a lack of data about the parameters used in the equations, (2) the data that are available are not representative of the particular instance being modeled, or (3) parameter values cannot be measured precisely or accurately because of limitations in measurement technology. Random, or sample, errors are a common source of parameter uncertainty that is especially critical for small sample sizes. More difficult to recognize are nonrandom or systematic errors that result from bias in sampling, experimental design, or choice of assumptions.

10.2.3.1 Waste Management Unit Parameters. As discussed in Section 4.4, existing databases were used to identify WMUs and as a basis for determining important emissions and dispersion model input parameter values. Landfills were characterized using EPA's Solid Waste (Municipal) Landfill Survey (U.S. EPA, 1988) for area and depth. Other parameters needed to characterize landfills were obtained from the *Industrial Subtitle D Survey* (Schroeder et al., 1987). The *Industrial Subtitle D Survey* (Schroeder et al., 1987) was also used to characterize surface impoundments, and TSDf survey data (U.S. EPA, 1987) were used to characterize treatment tank characteristics. Characterization of tanks was based on the *1986 National Survey of Hazardous Waste Treatment, Storage, Disposal, and Recycling Facilities* (U.S. EPA, 1987).

These databases were used to determine physical and operating characteristics for the WMUs modeled. The impact of the uncertainty associated with the information contained in these databases is unknown. There are several sources of this uncertainty, including age of the data, representativeness, missing data on waste volumes or capacity, multiple WMUs of the same type associated with a combined surface area and waste volume, accuracy of the reported data (i.e., measurement error), and limited information on WMU operating characteristics. Because some of these surveys were completed in 1987, uncertainty exists concerning changes in waste management practices since 1987. This is especially true for the tanks data; thus, the number of highly aerated tanks may be underestimated. Underestimation of the number of highly aerated tanks would result in lower emissions estimates and higher protective waste concentrations.

Source characterization also required making assumptions about the way WMUs are operated. For example, surface impoundments were assumed to be closed after 50 years and the site cleaned of all residual constituent contamination.

10.2.3.2 Infiltration Rates. The infiltration rate is a sensitive input parameter that directly impacts the flux of contaminant mass from a WMU to the subsurface. In this risk assessment, infiltration rates are obtained from two principal sources, depending upon the WMU and liner scenario considered: rates may be derived from sampled or representative WMU characteristics using one of three models; or infiltration rates may be selected from a distribution of rates founded on a sampling of available infiltration rate data. As a result, the uncertainty associated with this input parameter has components of both modeling and parameter uncertainty, depending on the particular setting.

Infiltration rates for unlined and clay-lined surface impoundments are derived at run time from WMU and site characteristics using Darcy's Law (see Section 4.4.2.4). Sources of uncertainty come from the site data as well as assumptions inherent in the conceptualization of variably saturated flow. Infiltration rates for unlined and clay-lined landfills are selected from a database of regional infiltration rates generated a priori using EPA's HELP model (see Section 4.4.1.5). Composite liner infiltration rates for both surface impoundments and landfills are based on liner performance data from a national survey of composite-lined WMUs (see Sections 4.4.1.5 and 4.4.2.4). For these values, uncertainty is propagated from the data collection, review and screening, and, in the case of surface impoundments, modeling uncertainty associated with estimating rates with the Bonaparte model (Bonaparte, et al., 1989). In all cases, risks may be under- or overpredicted due to uncertainty in the various models and data sources for this parameter.

10.2.3.3 Distribution Coefficients, K_d . The distribution coefficient, K_d , which is used in the source partition model, in the groundwater model, and in modeling constituent concentration in surficial soils, is an important parameter for modeling the fate and transport of metals in the environment. K_d values were calculated using MINTEQ in the groundwater model EPACMTP. However, in the source model, K_d values were derived from literature values. A comprehensive review of the literature was conducted to compile K_d data for an earlier rulemaking (U.S. EPA, 2000a). Despite this substantial earlier effort, considerable uncertainty remains in the literature-based values of K_d used in this analysis because data concerning K_d values for particular constituents reported in the literature were limited. In addition, reported values often were not accompanied by qualifying information. Conditions that affect K_d values (e.g., constituent concentration, metal species evaluated, pH, experimental technique) are often not reported in the literature, making interpretation of results difficult. For these reasons, substantial uncertainty concerning the values of K_d remains.

10.2.3.4 Chemical Degradation. The source models used in the dye and pigment listing consider loss of chemical mass due to biodegradation. This is a very important parameter for determining the amount of chemical that will ultimately leach and/or volatilize from a WMU. The values for biodegradation chosen for this analysis were predominantly from Howard et al. (1991). This reference provides a compilation of half-lives. The preferred data used by the authors were from experimental values; however, in some cases where experimental values were not identified, scientific judgments were made in order to estimate a value. For example, if only the value for aerobic degradation was found, the anaerobic value was sometimes estimated based on the aerobic value. In addition, the half-lives are presented as both a high and a low value. The values used in this analysis were for the high value, which is more conservative. The amount of biodegradation that occurs will also depend on various environmental parameters, including temperature, pH, and available biomass. Since experimental data are used to estimate biodegradation rates, these parameters are not specifically considered in the model. There is additional uncertainty in the results for chemicals for which no degradation data were available. For these chemicals, the values used for biodegradation were based on surrogates of similar chemical structure, leading to additional uncertainty in the modeling results for these chemicals.

10.2.3.5 Exposure Factors. For most exposure factors addressed, data analyses involved fitting distributions of data summaries from the EFH (U.S. EPA, 1997a,b,c), in most cases by fitting distributions to selected percentiles. It is assumed that little information is lost by fitting to percentiles versus fitting to raw data. However, some assessors believe that such analyses should always be based on raw data, synthesizing all credible sources. The data sets for time spent in shower clearly are affected by rounding and grouping of data. The fitting methods do not account for these sources of uncertainty.

Three standard two-parameter probability statistical distributions (gamma, lognormal, and Weibull) were used for this analysis. These distributions are special cases of a three-parameter distribution (generalized gamma) that contains them and allows for a likelihood ratio test of the fit of the two-parameter models. Other statistical distributions are possible (e.g., U.S. EPA, 2000b), but the technique used in this analysis offered considerable improvement over using a lognormal model in all cases and was appropriate for this analysis. In support of this conclusion, a comparison of results showed that the three-parameter generalized gamma

distribution did not significantly improve on goodness of fit over the two-parameter distributional forms in 58 of 59 cases at the 5 percent level of significance.

Although they offer significant improvement in objectivity over visual estimation, goodness-of-fit tests used to determine which statistical distribution to use for a particular parameter are themselves subject to some uncertainty that should be considered in their application to exposure factors. One area of concern is uncertainty about how the survey statistics in the EFH (U.S. EPA, 1997a,b,c) were calculated. All of the statistics that have been used to assess goodness of fit assume a random sample, which may or may not be a valid assumption for EFH data. Specifically, many of the EFH data sources are surveys that, in many cases, do not involve purely random samples. Rather, they use clustering and stratification, primarily for economic reasons.

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